MARINE TURTLE FISHERY OF CARIBBEAN NICARAGUA: HUMAN USE PATTERNS AND HARVEST TRENDS

By

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TABLE OF CONTENTS

Abstract of Dissertation Presented to the Graduate School
of the University of Florida in Partial Fulfillment of the
Requirements for the Degree of Doctor of Philosophy

MARINE TURTLE FISHERY OF CARIBBEAN NICARAGUA:
HUMAN USE PATTERNS AND HARVEST TRENDS

By

Cynthia Jean Lagueux

May 1998

Chairperson: Kent H. Redford, Ph.D.
Major Department: Wildlife Ecology and Conservation

The Miskitu Indian marine turtle fishery of Caribbean Nicaragua was studied to determine human use patterns and harvest trends in the fishery, and to evaluate its impact on marine turtle populations. Specific objectives of the study were to 1) characterize human use patterns, 2) quantify the number of animals harvested annually, 3) describe the size and sex of harvested animals, 4) describe the reproductive cycle of green turtles (*Chelonia mydas*), 5) evaluate the impact of the fishery on marine turtle populations, and 6) provide management and research recommendations.

Green turtles are targeted in the fishery, hawksbills (*Eretmochelys imbricata*) are harvested opportunistically, and loggerheads (*Caretta caretta)* and leatherbacks (*Dermochelys coriacea*) are captured incidentally in nets set for green turtles. Marine turtles are a source of protein, provide income through the sale of meat and tortoiseshell,

and most recently, used as bait in other fisheries. Current minimum annual harvest levels of green turtles are 10,000 to 11,000 animals, levels that are as high or higher than they have probably ever been for this coast. The majority of harvested green turtles are juvenile females. Like mature females, at least some of the mature males are not annual breeders. Large juveniles and adult hawksbills, loggerheads, and leatherbacks are also captured, although harvest levels are much lower than for green turtles.

Indications that green turtles are overharvested are the decrease in mean length and mass of harvested animals, and decrease in the mesh size of nets used. Population modeling studies of other long-lived, slow-maturing species indicate that increased mortality of juveniles and adults affect population growth the most. Results of this study do not indicate conclusively that marine turtle populations in Nicaragua are overharvested. Based on the magnitude of the green turtle fishery and its focus on large juveniles and adults, however, there is clearly cause for concern.

The development of a co-managed marine turtle fishery is recommended. Restrictions on the number, size, and sex of animals harvested, as well as, other recommendations to manage the fishery are made. Recommendations for future research on characteristics of marine turtle populations in the region are also made.

Resumen de la Disertación Presentada a la Escuela de Graduados
de la Universidad de Florida en Cumplimiento Parcial de los
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PESQUERÍA DE TORTUGAS MARINAS DE LA COSTA CARIBE
DE NICARAGUA: PATRONES DE USO HUMANOS
Y TENDENCIAS DE LA COSECHA

Por

Cynthia Jean Lagueux

Mayo 1998

Asesor Principal: Kent H. Redford, Ph. D.
Departamento: Ecología y Conservación de Vida Silvestre

Se estudió la pesquería de tortugas marinas entre los indígenas Miskitu en la costa caribe de Nicaragua, con fin de determinar los patrones de uso humano y las tendencias en la cosecha y para evaluar su impacto en las poblaciones de tortugas marinas. Los objetivos específicos del estudio fueron 1) caracterizar los patrones del uso humano, 2) cuantificar el número de animales cosechados anualmente, 3) describir el tamaño y sexo de los animales cosechados, 4) describir el ciclo reproductivo de las tortugas verdes (*Chelonia mydas*), 5) evaluar el impacto de la pesquería en poblaciones de tortugas marinas, y 6) dar recomendaciones para el manejo e investigación.

Las tortugas verdes son el blanco de la pesquería, mientras que las carey (*Eretmochelys imbricata*) son cosechadas en forma oportunista y los cabezones (*Caretta caretta*) y baulas (*Dermochelys coriacea*) son capturados incidentalmente en las redes

puestas para las tortugas verdes. Las tortugas marinas son una fuente de proteína, proporcionan ingresos por venta de la carne y el caparazón, y más recientemente son usadas como carnada en otras pesquerías. Los actuales niveles anuales mínimos de la cosecha de tortugas verdes son de 10.000 a 11.000 animales, probablemente los niveles más altos que hayan existido para esta costa. La mayoría de las tortugas verdes cosechadas son hembras juveniles. Igual que las hembras maduras, al menos algunos de los machos adultos no se reproducen anualmente. Los juveniles grandes y adultos de carey, cabezones y baulas también son capturados, aunque los niveles de cosecha son mucho más bajos que los de tortugas verdes.

Indicaciones de que las tortugas verdes son sobre cosechadas son la disminución en la longitud media y el peso de los animales cosechados y la disminución en el tamaño del ojo de las redes usadas. Estudios que usan modelos de población para otras especies de larga vida y maduración lenta indican que la mortalidad de juveniles y adultos es el factor que más afecta el crecimiento de la población. Los resultados de este estudio no indican en conclusión que las poblaciones de tortugas marinas en Nicaragua son sobre cosechadas. Sin embargo, con base en la magnitud de la pesquería de tortuga verde y su concentración en juveniles grandes y adultos, existe claro motivo de preocupación.

Se recomienda el desarrollo de una pesquería co-manejada de las tortugas marinas. También se recomiendan restricciones en el número, tamaño y sexo de los animales cosechados, así como otras sugerencias para el manejo de la pesquería. Igualmente se hacen recomendaciones para investigaciones futuras sobre las características de las poblaciones de tortugas marinas en la región.

Tanka plikanka ulbanka Kunhku kum Skul tnata alkan ra
Marikanka kum daukan sa pura luanka ai skul ka ra tanka Florida
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LIH MISKANKA NICARAGUA LALMA KABUKA RA
YUS MUNANKA TNATKA BARA ALKANKA TANKA

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Waild nani raiaka an watla bila kan kahbaia aslika

Miskitu nani lih yus munanka tnatkaba tanka pliki kaikan kan Nicaragua lalma kabuka unra, yus munanka tnanka bara alkanka tnatka ba wal tanka briaia dukiara, baku sin dia pitka kat sauhkanka brih auiaba lih aslika ra. Dia mihta nitkan naha tanka pliki kaikaia: 1) yus munan tnaka ba tanka briaia, 2) nahki pitka mani bani alkiba, 3) witin ai pawanka pitka alki ba tilara waihka baku mairin ba tanka kaikaia, 4) lih sahwanka tanka ba kau briaia dukiara, 5) dia pitka kat sauhki aula miskanka tnatka ba bui, 6) kupia kraukanka iaia yus munanka tnatka kum brih waia ba dukiara bakusin ai tanka pliki kaikaia dukiara.

Mamiskra nani brinka paliba lih (*Chelonia mydas*) sakuna axbil (*Eretmochelys imbricata*) sans ra alkisa, lagrit (*Caretta caretta*), bara lih siksa (*Dermochelys coriacea*) ba wal alkisa kuna lih tanka kahbuia ba tilara accident ra. Kabu lih ka ba aunhka ba

dukiara yus munisa, baku sin winka ba atki ilpka brisa, baku sin ai miskanka nani tnatka walara yus munisa. Lih kau wiria alkiba 10,000 wina 11,000 kat alkisa, naha na aima wala nani wal prakbia kaka nanara kau alkisa, baha pitka alkiba wina aihkika ba mairin tiara lupia kau alkisa, tila bara mairin aiapra kum kum alkisa, bara sin waihka nani kum kum ba mani bani sip alkras sa. Baku sin lagrit bara lih siksa wahma an almuk alkisa.

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CHAPTER 1
INTRODUCTION

Humans use natural resources for food, shelter, medicine, tools, pets, curios, barter, and as a source of income with which to procure other goods and services. Current and historical uses of natural resources for subsistence and trade have been well documented in the literature (e.g., Robinson and Redford 1991, 1994; Jorgenson 1993; Rose 1993, 1996; Bodmer 1994; Bodmer et al. 1994; Jenkins and Broad 1994; Bissonette and Drausman 1995; Jenkins 1995; Townsend 1995; Vincent 1996; Freese 1997). Use of sea turtles dates back to early humans with the discovery of what appear to be green turtle (*Chelonia mydas*) bones from excavations of Borneo caves (Harrisson 1962a, b, 1967), and tortoiseshell products from hawksbill turtles (*Eretmochelys imbricata*) dating back to the Han dynasty of China, beginning approximately 200 B.C. (Parsons 1972). The trade in hawksbill shell dates back to the 15th century B.C. (Parsons 1972).

Sea Turtles as a Resource

Sea turtles and their eggs have provided humans with a dependable resource for thousands of years. Nesting females and their eggs are highly vulnerable to harvesting, particularly the eggs, because they are an easy and relatively risk-free resource to exploit. Sea turtle eggs, as well as nesting females, can be a long-term, predictable resource for

humans because 1) turtles nest on their natal beaches, 2) females are iteroparous, 3) most species nest in relatively dense numbers, 4) nesting occurs seasonally (some species nest year-around at some locations), and 5) several species can nest on the same beach during different times of the year.

Unlike most other marine resources sea turtles can be kept alive, out of water, for weeks and thus provide humans with a dependable source of fresh meat for prolonged periods. Although nesting females and their eggs are more accessible for harvesting, animals can also be captured in the water, however, more skill and equipment are required. Some sea turtle species congregate on foraging grounds where they feed on sessile prey, assemble offshore of their nesting beaches, or are predictable in their migratory routes to and from the nesting beach. The concentration and predictability of animals at known in-water locations during various times of the year or during their lifespan makes them nearly as accessible to harvesting as nesting females and their eggs.

Worldwide Status of Sea Turtles

The current status of sea turtle populations worldwide is indicated by the threatened status of the seven extant species (IUCN 1996). Overharvest of animals and their eggs for human use, incidental capture, and habitat loss and degradation due to coastal development are some of the primary causes of worldwide population declines (e.g., Bjorndal 1982; National Research Council 1990; Eckert 1995; IUCN 1997; Lutcavage et al. 1997). Currently, the international sale of sea turtles and their products is illegal among the 142 (as of September 1997, Anon. 1997) signatory nations of the

Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES). All seven species of sea turtles are listed under Appendix I, species threatened with extinction (CITES 1992).

Historical Harvest of Sea Turtles, Except Nicaragua

The harvest of sea turtles for their meat, shell, oil, and calipee, and their eggs has occurred for thousands of years, probably since hominids encountered female turtles, their eggs, and hatchlings on coastal beaches. Human impact on turtle populations prior to large-scale commercialization is unknown, however, population declines are well-documented once exploitation for commercial purposes began (Ingle and Smith 1949; Carr 1954; Parsons 1956, 1962, 1972; Rebel 1974; Cato et al. 1978; Bjorndal 1982; Dodd 1982; King 1982; Milliken and Tokunaga 1987; Meylan 1997a). Although the eggs of all seven sea turtle species are in demand, green and hawksbill turtles have received the greatest pressure in both numbers of animals harvested and duration of exploitation (Hornell 1927; Ingle and Smith 1949; Carr 1954; Parsons 1956, 1962, 1972; Freeman-Grenville 1962; Rebel 1974; Cato et al. 1978; Bjorndal 1982; King 1982; Milliken and Tokunaga 1987; Meylan 1997a). Tortoiseshell from hawksbill turtles was among the items in demand and sought after by early civilizations (Parsons 1972). It carried prestige and indicated wealth among the upper class. Cleopatra's bathtub was reportedly made from hawksbill shell (Parsons 1972). Throughout various periods in history and for shorter periods of time, the loggerhead (*Caretta caretta*), olive ridley (*Lepidochelys olivacea*), Kemp's ridley (*Lepidochelys kempi*), leatherback (*Dermochelys coriacea*), and

flatback (*Natator depressus*) turtles have been in demand for their skin, oil, and meat (Bjorndal 1982; Ross 1982). In addition to direct take, in-direct human-induced mortalities also occur (National Research Council 1990; Lutcavage et al. 1997).

It is necessary to have, at least, a general awareness of the extent of historical and contemporary levels of direct and in-direct take on sea turtle populations to avoid the shifting baseline syndrome. This syndrome occurs when scientists view the baseline of stock size or species composition from the start of their careers and not from a historical perspective (Pauly 1995; Sheppard 1995). In addition, knowledge of the historical decline of sea turtle populations worldwide provides a perspective with which to view current population levels. Thus, the following section is a compilation of historical sea turtle exploitation worldwide, except for Nicaragua and resulting population declines by species and geographic region. The exploitation of sea turtles from Nicaragua will be presented later.

Mediterranean and East Atlantic Regions

Due to overexploitation, marine turtle populations off the coast of Israel and Turkey have declined dramatically since the end of World War I (WWI) (Sella 1982). At the peak of the season in the mid-1930s, an estimated 30,000 green and loggerhead turtles were captured offshore of northern Israel (Sella 1982). Hornell (1934 cited in Sella 1982) reported the export of 2,000 turtles a year from Palestine to Egypt. Between 1952 and 1965, up to 15,000 green and loggerhead turtles were harvested off the Turkish coast, processed and the product sent to Europe (Sella 1982). Due to a decline in harvest rates

by the mid-1960s the center of the fishery moved to another location along the coast and by 1965 more than 10,000 turtles, mostly greens, were captured (Sella 1982).

By 1967, in Madeira, an estimated 1,000 loggerheads were killed annually, primarily for human consumption and as wall hangings for the tourist trade (Brongersma 1982). By 1979, an estimated 2,000 loggerheads were killed annually and the primary market had shifted to the tourist industry (Brongersma 1982).

In 1835, on Ascension Island, up to 40 or 50 green turtles were harvested from the nesting beaches in a single night with an annual harvest of more than 2,500 animals (Anonymous cited in Parsons 1962). In the 1840s and 1850s, between 600 and 800 green turtles were harvested annually (Colonial Office Reports cited in Parsons 1962). By the 1920s, an average of 60 turtles a year were exported and by 1932 no mention of exports were reported (Colonial Office Reports cited in Parsons 1962).

West Indian Ocean Region

Sea turtle populations in Tanzania have probably been reduced since prehistory (Frazier 1982a). From the late 1800s until recently, Zanzibar was a major exporter of tortoiseshell, however, the scutes originated from throughout the Indian Ocean and were exported to Europe and Asia via Zanzibar (Frazier 1982a, b). Two thousand years of exploitation have decreased the number of green and hawksbill turtles on the Kenya and Somalia coasts (Parsons 1962; Frazier 1982a). In 1951, a turtle soup cannery was opened in Kenya, and by 1954, an estimated 200 turtles were captured annually to supply the cannery (Parsons 1962). From 1954 to 1959, 1,000 to 1,500 live turtles were exported

annually from Kenya to England (Parsons 1962). In Mauritius, sea turtles no longer nest due to overexploitation (Frazier 1982b; Hughes 1982). In Madagascar, only the hawksbill is exploited for export, however, the green, loggerhead, olive ridley, and leatherback turtles are exploited mainly for local consumption (Hughes 1982). Tortoiseshell has been an important export for Madagascar from 1613 to the early 1970s (Hughes 1975). From the mid-1800s to the mid-1900s, a minimum of 1,600 adult hawksbills were harvested annually (Hughes 1973, 1975). By WW I, tortoiseshell exports declined sharply and by the mid-1970s annual exports were around 250 kg (approximately 100 animals) (Hughes 1975). Based on a survey of the southwest coast of Madagascar, Hughes (1971) calculated that over 13,000 turtles were harvested annually, 50% of the harvest comprised green turtles and the remainder of the catch distributed approximately equally among loggerheads, hawksbills, and olive ridleys.

The harvest of green and hawksbill turtles from the Republic of Seychelles has occurred since its discovery by Europeans in 1609 (Parsons 1972; Frazier 1982a; Mortimer 1984; Stoddart 1984) and the decline of green turtles probably began in the late 1700s (Mortimer 1984) and hawksbills by the 1860s (Hornell 1927). In 1780, 454 to 907 kg of hawksbill shell were harvested, and from 1893 to 1925, 42,727 kg of tortoiseshell ($\approx$ 30,500 animals) were exported (Hornell 1927). From 1893 to 1968, Stoddart (1984) reports the export of 60,780 kg of tortoiseshell ($\approx$ 45,000 animals).

Green turtles in the Seychelles have been in demand for their meat, calipee, oil, eggs, blood (drunk as a health tonic), shell, cawan (primarily plastron but also carapace scutes), and bones (Hornell 1927; Frazier 1975; Mortimer 1984). As early as 1860,

concern was expressed over the enormous numbers of green turtles killed for their cawan (Hornell 1927). The most drastic decline in green turtles began in the early 20th century (Hornell 1927; Mortimer 1984). From 1907 to 1909, Hornell (1927) reported approximately 20,500 green turtles were harvested. Between 1923 and 1925, approximately 14,000 green turtles were harvested for only their calipee (cartilaginous tissue located between the belly plates) and the remains of the animals were left on the beach to rot (Hornell 1927; Parsons 1972). Indications that the population in the Seychelles was in decline was the decrease in nesting females, decrease in the number of animals harvested, and a decline in the heaviest animals (Hornell 1927).

Asian Region

Between 4,000 and 5,000 sea turtles were captured annually in the southern Indian state of Tamil Nadu, the time period was not given (Kar and Bhaskar 1982). Of 5,000 olive ridley egg clutches laid along a 50 km stretch of beach in Tamil Nadu, approximately 90% were harvested by humans or predated by *Canis* spp. (Whitaker 1977 cited in Kar and Bhaskar 1982). Prior to 1975, the state government of Orissa sold permits to collect approximately 2 million olive ridley eggs/yr ($\approx$ 100 eggs/clutch), which were sold locally, regionally, and in Calcutta (Bustard 1980; Kar and Bhaskar 1982). During a three month period, in 1978/1979, West Bengali fishers harvested over 21,000 olive ridleys in front of the Orissa nesting beach (Biswas pers. com. to Kar and Bhaskar 1982). During the 1981/1982 nesting season, an estimated 90,000 olive ridleys were harvested, and during the 1982/1983 season, an estimated 10,000 ridleys were landed in

West Bengal and transported inland for sale (Silas et al. 1983). Due to overharvest for tortoiseshell, hawksbills have been extirpated from the south coast of Sri Lanka (Frazier 1982b).

At one time, five species of sea turtles were found in Thailand, and all were exploited for their eggs. From the period 1963 - 1966 to the period 1972 - 1973, a 70% decrease occurred in the number of eggs collected annually at one site (Polunin 1975; Settle 1995). Today, green, hawksbill, olive ridley, and leatherback populations are seriously reduced and the loggerhead is believed to be extirpated (Polunin 1975; Mortimer 1988; Settle 1995).

The Rantau Abang leatherback rookery in the State of Terengganu, Malaysia, is well known for its once large numbers of nesting leatherback turtles. Today, they are critically endangered. In the late 1950s, an estimated 2,000 females laid approximately 10,000 egg clutches annually (Mortimer 1990). Since the 1950s, the nesting population has declined steadily and catastrophically. A 1978 survey reported egg yields from leatherbacks had declined 34% since 1956 (Siow and Moll 1982). In 1989, fewer than 200 egg clutches were laid, a 98% decrease in nesting activity (Mortimer 1990; Chan and Liew 1995). In recent years, no more than 20 females nested annually (Mortimer pers. com.). During the 1989 nesting season, Mortimer (1990) reports that as many as 1,000 tourists viewed a single nesting female. This catastrophic decline in nesting females is attributed to a combination of factors, perhaps the most important being the nearly 100% harvest of leatherback eggs laid during most of the present century but also adult mortality caused by entanglement in fishing gear.

In addition to the leatherback rookery in the State of Terengganu, there is also green, hawksbill, and on occasion olive ridley nesting (Siow and Moll 1982; Mortimer 1991). Between 1956 and 1978, green turtle egg production at Terengganu has declined between 43% and 57%, (Siow and Moll 1982; Limpus 1994, 1997). The largest concentration of green and hawksbill nesting in Terengganu occurs on Pulau Redang Island (Mortimer 1991). Prior to 1984, nearly 100% of the green and hawksbill eggs laid were collected for human consumption (Mortimer 1990). Since the late 1950s, declines in the number of green, olive ridley, and hawksbill eggs laid in Terengganu ranged between 52% and 85% (Mortimer 1990).

In 1839, on Talang Talang Kechil Island, Sarawak (one of three Sarawak islands known for green turtle nesting) five to six thousand green turtle eggs were collected every morning, the duration of this harvest rate was not reported (Hendrickson 1958; Keppel 1847 cited in Harrisson 1962a). By the mid-19th century, sections of beach were leased by the government and nearly 100% of the green turtle eggs laid were collected (Harrisson 1951, 1962b; Hendrickson 1958). Since 1927, there has been a > 90% decline in egg production on Sarawak's turtle islands (Harrisson 1962b; Limpus 1994, 1997).

On Sabah, from 1947 to 1978, there has been more than a 50% decline, from 706,960 eggs (≈ 6,670 clutches) in 1947 to 322,102 eggs (≈ 3,039 clutches) in 1978, in the number of green turtle eggs laid (Harrisson 1964, 1966, 1967; de Silva 1982; Limpus 1994). From 1951 to 1985, the green turtle nesting population of the Sulu Sea Turtle Islands of Sabah, Malaysia/Philippines has declined > 75% and possibly as much as 90% (Limpus 1997).

In Indonesia, between 1975 and 1978, an average 7,531 ± 1,275 turtles/yr (species not reported) were exported to Japan, Singapore, and the United States (Polunin and Sumertha Nuitja 1982; Suwelo et al. 1982). An estimated 10,000 juvenile hawksbills were captured annually from Sulawesi, and from 15,000 to 20,000 juvenile hawksbills were captured annually from Sumatra, duration of the harvest was not given (Kajihara 1974 cited in Polunin and Sumertha Nuitja 1982). From the Java, Flores, and Banda seas, approximately 5,000 adult hawksbills were captured annually prior to 1971 and 30,000 adults were captured annually after 1972 (Kajihara 1974 cited in Polunin and Sumertha Nuitja 1982). During 1988 and 1989, over 1.5 tones of bekko (Japanese term for hawksbill shell) were exported in violation of CITES to Hong Kong, Taiwan, and Singapore (Barr 1992). From 1934 to 1984, green turtle egg production declined more than 80% (Limpus 1994). Barr (1992) estimated 7 to 9 million sea turtle eggs are harvested annually in Indonesia, essentially 100% of the eggs laid.

One of the largest landings of green turtles in the world occurs in Bali (Barr 1992; Limpus 1994, 1997). By 1950, local sea turtle populations around Bali were seriously depleted (Sumertha Nuitja 1974 cited in Polunin and Nuitja 1982). From 1968 to 1970, Sumertha Nuitja (1974 cited in Polunin and Nuitja 1982) reported the consumption of 28,800 turtles in two districts of Bali. In 1973, a Balinese company exported 5,000 to 6,000 stuffed sea turtles and leather from an additional 3,000 turtles each month (Polunin 1975). More than 30,000 turtles harvested annually from throughout the Indonesian archipelago were landed in Bali at the height of the trade, no date was provided (Barr 1992; Limpus 1997). In the late 1980s, the World Wildlife Fund estimated the total

harvest of green turtles in Indonesia at 50,000 (Limpus 1997). In 1990, Greenpeace investigators reported at least 21,000 sea turtles landed in southern Bali, although the Indonesian government claimed harvest levels had decreased to approximately 10,000 to 15,000 animals annually (Barr 1992). Currently, 25,000 sea turtles, mostly large green turtles, are imported annually into Bali from throughout Indonesia (Limpus 1997). In 1994, several thousand hawksbills, harvested from throughout Indonesia, were landed in Bali (Limpus 1997).

Since 1956 in Irian Jaya, there has been a > 90% decline in nesting leatherback turtles (Limpus 1994). Approximately 60% of the leatherback eggs laid on the world's third largest leatherback nesting beach (north coast) were collected for local consumption and sale (Starbird and Suarez 1994). In addition, approximately 200 leatherbacks are currently harpooned off the southwest coast by local villagers for consumption (Suarez and Starbird 1995). The annual harvest of green turtles in Papua New Guinea is estimated between 10,000 and 20,000 animals (Limpus 1997). Apparently due to overharvesting sea turtles are no longer found feeding offshore or nesting on beaches in the vicinity of villages in Papua New Guinea (Spring 1982).

In 1953, over 1 million eggs were harvested from the Philippine Turtle Islands (Parsons 1962). Kajihara (1974 cited in Polunin 1975) reported 5,000 adult hawksbill and 50,000 large green turtles were captured annually in the Sulu Sea, and between 1961 and 1972, tortoiseshell from approximately 45,000 hawksbill were exported to Japan. From 1951 to 1984, there has been a > 75% decline in green turtle egg production from the Philippine Turtle Islands (Limpus 1994).

In Japan, between 1880 and 1890, 1,000 to 1,800 green turtles were harvested yearly from Ogasawara Island and by the mid-1920s harvest rates had declined to fewer than 250 animals. Since 1973, when the Japanese regained possession of the island, harvest rates have been between 45 and 225 green turtles/yr (Horikoshi et al. 1994). In addition, Japan has been the largest importer of sea turtle products in the world with imports during the last 20 years representing over two million animals (Barr 1992).

Australian Region

In the 1920s at least two turtle processing factories operated on Northwest Islet and one on Heron Island, Australia (Parsons 1962). During the 1924/1925 season, approximately 1,600 green turtles were processed (Parsons 1962). During the 1928/1929 season so few nesting females were available on Heron Island that nesting animals on nearby islands were harvested (Moorhouse 1933). For 40 years, prior to 1954, green turtles were harvested on the Capricorn Reef, transported to Brisbane and shipped to England (McNeill 1955 cited in Parsons 1962). On the west coast of Australia, approximately 50 animals/wk were processed in a local turtle processing plant until 1951 (Caldwell 1951 cited in Parsons 1962). Although sea turtles are now protected in Australia, indigenous people in Queensland and Western Australia are allowed to take turtles for their own use (Limpus 1982). It is estimated that 10,000 green turtles are harvested annually from the Torres Strait, of these approximately 4,000 are harvested by Torres Strait Islanders and used locally, and the remainder are harvested by Papua New

Guineans and sold in their coastal markets (Daly 1990; Limpus 1982). Nearly 100% of the eggs laid near indigenous communities are harvested (Limpus 1982).

Central Pacific Ocean Region

Coastal inhabitants throughout the Central Pacific Ocean have harvested marine turtles for thousands of years. Marine turtle populations throughout the islands have declined within historical times (Balazs 1982). In the Caroline Islands, the harvest of marine turtle eggs is uncontrolled and occurs on all the islands in the group (McCoy 1982). Between 1985 and 1989, scutes from approximately 22,300 hawksbills were exported to Japan from the Solomon Islands and Fiji (Daly 1990). Prior to mid-1994, an estimated 2,000 hawksbills/yr were harvested from foraging grounds in Fiji (Limpus 1997).

Eastern Pacific Ocean Region

Mexico began the commercial exploitation of olive ridleys in 1961 (Márquez et al. 1976; Cato et al. 1978). Both Mexico and Ecuador exported large quantities of olive ridley and Pacific green turtle skins and leather to Japan, France, Spain, Italy and the United States (Pritchard 1978; Mack et al. 1982; Milliken and Tokunaga 1987). From 1948 to 1956, approximately 60 tones of olive ridleys/yr were harvested and in the early 1960s, between 250 and 500 tones/yr were harvested. During the peak period of exploitation, between 1965 and 1969, over 30,000 tones/yr of olive ridleys were harvested (Márquez et al. 1976), representing approximately 700,000 animals (Márquez unpubl. data cited in Cliffton et al. 1982). However, according to Carr (1972), the

Mexican government underestimated the total catch, he estimated more than a million olive ridleys were harvested in 1968 alone. The harvest of olive ridleys had been so intensive that at Piedra de Tlacoyunque, one of only four Pacific coast Mexican *arribada* (Spanish for mass arrival of females on a nesting beach) nesting beaches, the aggregation of turtles had been reduced from 30,000 to only a few hundred between 1968 and 1969 (Carr 1972; Pritchard 1979). By the early 1970s, three of the four Mexican olive ridley *arribada* populations had been destroyed and the remaining location of mass nesting, Playa Escobilla, Oaxaca, was being severely exploited (Carr 1967, 1979; Frazier 1981).

In 1977 and 1978, an estimated 70,000 and 58,000 olive ridleys were harvested from Playa Escobilla, respectively (Cliffton et al. 1982). The decrease in size and number of *arribadas* that occurred at Playa Escobilla indicated that the nesting population was overharvested (Cliffton et al. 1982), however, the harvest continued. From 1980 to 1985, the average annual harvest of olive ridleys was 32,343 (Hernandez M. and Elizalde A. 1989 cited in Rose 1993). Harvest quotas were reduced from 48,944 in 1980 to 23,000/yr between 1986 to 1990 (Peñaflores S. and Nataren E. cited in Rose 1993). In 1989, 2 to 12 boats illegally harvested 80 to 600 turtles/day in front of Playa Escobilla (Blanco-Casillo 1990). In May 1990, Mexico declared a permanent ban on all harvest and trade in sea turtles and their products (Aridjis 1990; Rose 1993).

At the turn of the century, an estimated 1,000 green turtles/mo were harvested from Baja California and sent to San Diego, California (O'Donnell 1974). Green turtles once nested along the coasts of Nayarit, Sinaloa; southern Sonora; and Baja California, Mexico; however, today only the olive ridley nests in these areas (Felger and Cliffton

1977). The increase in human population and subsequent exploitation of nesting green turtles are blamed for the disappearance of these northern Pacific Mexico nesting populations (Felger and Cliffton 1977). As larger turtles became more scarce in the late 1960s, juvenile greens were harvested at a rate of approximately 250 to 360 turtles/day (Cliffton et al. 1982). In the Gulf of California, for a three to four month period in the winter of 1975, five turtle boats harvested an estimated 140 to 175 green turtles/wk as the turtles lay dormant in the mud (Cliffton et al. 1982). The number of turtles had declined so drastically by 1975, Cliffton et al. (1982) reported that it took five boats of fishermen with diving gear to capture as many turtles as one boat of Seri Indians with harpoons in the 1960s. By 1977, Mexican fishermen had destroyed the population of dormant green turtles in the Gulf of California (Felger and Cliffton 1977).

Green turtles also once nested along the southern Pacific coast of Mexico; however, today only one major nesting site remains, Colola-Maruata Bay, Michoacán (Cliffton et al. 1982). Nahuatl Indian informants reported there were 10 to 20 times more nesting green turtles in 1970 than in 1979 (Cliffton et al. 1982). Over 4,500 metric tons of green turtles were landed on the Pacific coast of Mexico (Márquez et al. 1976), estimated to represent approximately 125,000 adult and subadults (Cliffton et al. 1982). In the early 1970s, the Nahuatl Indians estimated they harvested 15,000 to 20,000 eggs/night at Maruata Bay and 70,000 eggs/night at Colola (Cliffton et al. 1982). In 1978, at least 10,500 green turtles were legally harvested from Michoacán and Jalisco (A. Suárez pers. com. to Cliffton et al. 1982). In 1979, although a closed season had been

established, approximately 3,000 green turtles were illegally harvested (Cliffton et al. 1982).

In 1685, near Coiba Island, Panama, Dampier reported harpooning sea turtles every day (Parsons 1962). Reports of abundant numbers of turtles continued in 1741 and 1794. By 1956, however, only a few turtles were reported (Parsons 1962). During the 1970s, marine turtle populations declined drastically (Cornelius 1982). From 1964 to 1976, over 96,000 kg of hawksbill shell, representing approximately 55,670 animals, were officially exported from Panama (Vallester 1978 cited in Cornelius 1982).

In Ecuador, during the 1970s, at least six companies were involved in exporting frozen sea turtle meat for human consumption and salted skin for the leather trade (Green and Ortiz-Crespo 1982). From 1970 to 1978, up to 90,000 olive ridleys/yr were processed and exported. For 1977 alone, Japan imported 66% and Italy imported 25% of the skins exported from Ecuador (Green and Ortiz-Crespo 1982). The majority (72%) of the meat was imported by the United States (Green and Ortiz-Crespo 1982).

In Peru, from 1965 to 1985, the mean annual turtle harvest was estimated at 1,222 $\pm$ 1,636, and in 1987, it was estimated at > 22,200 animals (Aranda and Chandler 1989). The harvest most likely represented a combination of greens, olive ridleys, and leatherbacks, although the species harvested was not provided. The decrease in mesh size of the nets used in the fishery from a 59 cm bar in 1979 (Hays Brown and Brown 1982) to a 25 cm bar in the early 1990s (Vargas et al. 1994) could indicate a decrease in the mean size of animals due to overharvest.

Eastern United States Region

At least four species of sea turtles were harvested along the east, Gulf of Mexico, and west coasts of the United States. From 1880 to 1947, a minimum 1.5 million kg of sea turtles or ≈ 10,700 animals (estimated number of animals based on $\bar{x} = 136.2$ kg/green turtle (National Research Council 1990)), principally greens, were landed in Florida (Ingle and Smith 1949; Rebel 1974). From 1950 to 1971, a minimum 509,500 kg of green turtles or ≈ 3,740 greens (estimated number of animals based on the same $\bar{x} = 136.2$ kg/green turtle), and from 1951 to 1971, a minimum 66,535 kg of loggerheads and Kemp's ridleys combined or from ≈ 590 - 1,665 animals (estimated number of animals based on $\bar{x} = 113$ kg/loggerhead or $\bar{x} = 40$ kg/Kemp's ridley (National Research Council 1990)), were landed in Florida (Rebel 1974). From 1890 to 1976, a minimum 4.3 million kg of sea turtles (ranging from ≈ 31,600 to 38,100 animals based on mean mass of green or loggerhead turtles as described above) were landed at eight U.S. states and territories (Witzell 1994). An estimated 11,000 turtles (loggerheads, Kemp's ridleys, and greens) were unintentionally killed annually on the United States east coast and Gulf of Mexico by shrimp trawlers (Henwood and Stuntz 1987). In the United States, indirect human-induced mortality of loggerheads was estimated at 5,550 to 55,500 animals annually and for Kemp's ridleys it was estimated at 555 to 5,550 animals annually. This mortality was caused by shrimp trawls, discarded fishing gear and debris, other fisheries, dredging, collisions with boats, oil-rig removal, and electric power plants (National Research Council 1990).

Gulf of Mexico Region

Up to the mid-1950s, as many as 2,000 nesting green turtles/yr were harvested from the Yúcatan Peninsula, Mexico and exported to the United States (Parsons 1962). Between 1949 and 1969, > 3.6 million kg ($\bar{x}$ = 173,000 kg/yr) of green turtles and > 800,000 kg ($\bar{x}$ = 47,800 kg/yr) of loggerheads were harvested annually (Rebel 1974). Annually, 200 kg of hawksbill shell was harvested from the Yúcatan, no time period is provided (Carranza 1967 cited in Rebel 1974). During the mid-1970s, the established quotas for the east coast of Mexico ranged from 420 turtles to 2,000 turtles/yr, divided evenly between greens and loggerheads (Cato et al. 1978). Populations of four of the five species (hawksbill, loggerhead, green, and Kemp's ridley) of sea turtles that occur on the east coast of Mexico have declined (Hildebrand 1982). The Kemp's ridley is the most endangered of the seven species of sea turtles (Ross et al. 1989; Pritchard 1997). Decline in the nesting population began prior to 1966 with high levels of egg exploitation (Pritchard and Márquez 1973, Ross et al. 1989, Márquez 1994), and continues today due to incidental capture by shrimp trawlers (Ross et al. 1989; National Research Council 1990; USFWS/NMFS 1992).

Greater Caribbean Region

In the greater Caribbean, sea turtles played an important role in the expansion and dispersal of Europeans to the New World during the period of discovery, conquest, and colonization (Carr 1954; Parsons 1962). Carr (1954) credited the green turtle as being the single most important dietary factor that supported the opening up of the Caribbean to

European colonization. The green turtle provided ship crews with a source of fresh meat and allowed for extended periods of travel (Carr 1954; Great Britain Colonial Office Reports 1929 cited in Parsons 1962). Because of overexploitation, however, many nesting and foraging populations throughout the greater Caribbean were depleted or extirpated during early European expansion (Carr 1954; Parsons 1962, 1972; Dodd 1982; King 1982).

Parsons (1962) suggests that commercial turtling in the west Atlantic probably began in Bermuda, where at one time, there was a large assembly of nesting and foraging green turtles (Ingle and Smith 1949; Parsons 1962). However, in spite of legislation established in 1620 to protect sea turtles, within 150 years of English settlement sea turtle populations around Bermuda were so reduced that a commercial harvest was no longer profitable (Garman 1884b cited in Carr 1952; Parsons 1962). Carr (1954) suggests that Bermuda was probably the first documented green turtle rookery to be extirpated. In 1671, Bahamian officials were asked to prepare legislation that would protect green turtles against overexploitation, however, no action was taken (Great Britain Public Record Office 1889 cited in Parsons 1962). By the 1700s, the Bahamian green turtle population was also destroyed (Carr 1954; Dr. Archie Carr (interview) 1984).

The Cayman Islands were once known for the size of their green turtle rookery, which supported the largest turtle fishery in the New World (Lewis 1940; Carr 1954; Parsons 1962; King 1982). As early as 1503, Columbus recorded the massing of turtles around the Cayman Islands (Carr 1954; Morison 1942 cited in Parsons 1962) and Long (1774 cited in Lewis 1940) described how during the nesting season there were so many

turtles migrating towards the Cayman Islands that lost ships would navigate by the sound of their swimming towards the Caymans. For almost 200 years, sailing ships from many nations (e.g., British, Dutch, and French) arrived each summer to "turn turtle" (while on the nesting beach female turtles are turned over on their backs to prohibit them from returning to the water) (Parsons 1962). In 1684, it was reported that approximately 2,000 inhabitants of Port Royal, Jamaica, as well as, an unknown number of inland inhabitants, fed daily on green turtle meat (Molesworth cited in Parsons 1962). By 1802, a little less than 150 years after English settlement had begun, green turtle populations had become so depleted that Cayman turtlers sailed first to the south coast of Cuba, then the Gulf of Honduras, and finally to the Miskito coast of Nicaragua in search of ever dwindling stocks of turtles (Lewis 1940; Carr 1954; Parsons 1962; King 1982).

Historical Harvest of Sea Turtles from Caribbean Waters of Nicaragua

Green Turtles

Turtles have been harvested from Nicaragua's coastal waters and beaches for at least the past 400 years (Carr 1954; Parsons 1962; Roberts 1965; Dampier 1968; Nietschmann 1973; Mortimer 1981; Montenegro Jiménez 1992; Lagueux 1993). Unfortunately, no information is available on harvest rates prior to European arrival. However, as early as 1633, the English had established a trading station at Cabo Gracias a Dios (near the Honduras/Nicaragua border) (Parsons 1962). Parsons (1962) speculated that the Miskitu Indians taught the English and their colonists how to turtle. By 1722, Jamaican and possibly Cayman boats were annually visiting the Miskito Cays of

Nicaragua to catch and purchase green turtles and hawksbill shell from the Miskitu Indians (Fernández cited in Parsons 1962). However, turtling by the Cayman Islanders off the coast of Nicaragua did not occur with any regularity until the early 1800s (Lewis 1940; Parsons 1962). Simmonds (cited in Parsons 1962) reported that by 1878, up to 15,000 turtles, although the species landed was not stated the majority were probably green turtles, annually were landed in Europe, most of them having been caught by the Cayman fleet which was known to harvest turtles in Nicaraguan waters.

During the first-half of the 20th century approximately 2,000 to 4,000 green turtles were harvested annually from the Nicaragua coast by Cayman turtlers (Ingle and Smith 1949; Parsons 1962). By the mid-1960s, after several hundred years of exploitation, the Nicaraguan government no longer permitted Cayman Islanders to turtle within their waters (Nietschmann 1973, 1976). Apparently, the Nicaraguan government was not motivated by its concern for turtle conservation, but by its interest in securing a constant supply of turtles for their newly established turtle processing plants (Rainey and Pritchard 1972) and to decrease competition with other countries on the international market. In late 1968, the first of three Nicaragua marine turtle packing plants began processing green turtles for export (Nietschmann 1973, 1974). From 1966 to 1976, Nicaragua exported 445,500 kg (equivalent to approximately 10,000 animals) of sea turtle products into the United States alone during 7 of these 10 years (Cato et al. 1978). From 1969 to 1976, up to 10,000 green turtles were harvested annually from the offshore waters of Nicaragua for local and foreign consumption (Nietschmann 1972, 1973). By 1977, the processing plants were closed and Nicaragua became a signatory of CITES

(Hemley 1994). From 1985 to 1990, the sale of 16,700 green turtles was recorded in the Puerto Cabezas, Nicaragua market (Montenegro Jiménez 1992).

Hawksbill Turtles

Hawksbills have been harvested from the offshore waters of Caribbean Nicaragua and from the nesting beaches of the mainland and offshore cays for probably as long as green turtles have been harvested from this region. Annual boat trips by the Miskitu Indians to southern Nicaragua, Costa Rica, and to northern Panama to harvest hawksbills is reported for as early as the 1600s (Parsons 1972; Nietschmann 1973). In the mid-18th century, annual exports of tortoiseshell to Europe averaged 6,000 to 10,000 lbs (2,722 - 4,536 kgs) (Parsons 1956, 1972). The shell was traded by the Miskitus to the English for cloth, guns, rum, and other goods (Parsons 1956, 1972; Nietschmann 1973). During a 12-mo period beginning in October 1968, 41 hawksbills were harvested by one Miskitu Indian village (Nietschmann 1972, 1973). For the first six months of 1969 compared to the same time period in 1971 the harvest of hawksbills by one village increased almost 400%, from 27 to 107 animals (Nietschmann 1972, 1973). During the early 1970s, approximately 1,000 to 1,200 hawksbills were harvested annually and exported to Japan (Nietschmann 1981). Based on Japanese customs statistics, from 1970 to 1986, Nicaragua exported 14,519 kg of tortoiseshell to Japan, representing approximately 13,000 hawksbills (Milliken and Tokunaga 1987). Although Nicaragua has been a signatory of CITES since 1977 (Hemley 1994), approximately 20% of this trade occurred post-1977 (Milliken and Tokunaga 1987).

Loggerhead and Leatherback Turtles

Very little is known about the loggerhead in Nicaraguan waters. Unconfirmed reports indicate that loggerheads nest infrequently on Nicaragua's Caribbean coast (Carr et al. 1982). Animals are captured incidentally in nets set for green turtles. Although loggerhead meat is not eaten, throat and shoulder skin from loggerheads, as well as, green and hawksbill turtles was exported to Europe (Nietschmann 1972, 1981; Bacon 1975). Leatherback turtles are found in Nicaraguan waters and possibly nest in low numbers on the mainland (Bacon 1975; Carr et al. 1982; this study). Prior to this study, nothing was known about their capture from Nicaragua's Caribbean waters.

Marine Turtles and the Nicaragua Fishery

Today, the largest remaining foraging population of green turtles in the Atlantic Ocean is located in coastal waters of eastern Nicaragua (Carr et al. 1978). The extended continental shelf found in this region comprises cays, coral reefs, and extensive seagrass beds. Green turtles use this area for foraging, developmental habitat, and as a migratory pathway to the Tortuguero, Costa Rica nesting beach. Data from international tag recoveries demonstrate that green turtles tagged in the Bahamas, Bermuda, Costa Rica, Cuba, Florida, Grand Cayman, Mexico, Panama, and Venezuela have been captured in Nicaragua's offshore waters (Solé 1994; Bjorndal and Bolten 1996; Bagley *in litt.*; Bresette *in litt.*; Ehrhart pers. com.; Lagueux pers. obs.; Meylan *in litt.*; Moncada pers. com.).

Marine turtles and their products no longer are legally exported from Nicaragua, however, Miskitu and Rama Indians continue to conduct a legal marine turtle fishery for local consumption centered on the green turtle. In 1965, the Nicaragua government established a closed season to protect sea turtles in their Caribbean waters for several months/yr, although turtles occur year around (Nietschmann 1972, 1973; Rebel 1974; Weiss 1976; Bacon 1981; Peralta Williams 1991; Montenegro Jiménez 1992; pers. obs.). Because the law is not enforced, and some have argued, unenforceable (Nietschmann 1973; Peralta Williams 1991; D. Castro pers. com.), turtles continue to be harvested during the closed season.

Hawksbill turtles are opportunistically captured by lobster divers, in nets set for green turtles, and while nesting on mainland beaches or offshore cays. Juvenile and adult hawksbills tagged in at least three countries (Costa Rica, Mexico, and the U.S. Virgin Islands) throughout the greater Caribbean have been harvested in Nicaraguan waters (Carr et al. 1966; Carr and Stancyk 1975; Bjorndal et al. 1985; Hillis 1994; Garduño *in litt.*; see Meylan 1997b for review). Green and hawksbill turtle meat are used for subsistence and green turtle meat is sold in local markets. Hawksbill shell is sold to local artisans who fashion various jewelry items that can be found for sale throughout the country.

Loggerhead and leatherback turtles are not targeted in the fishery, however, they are also captured in nets set for green turtles. Two loggerheads, one tagged in Panama (Meylan *in litt.*) and one in the Azores, Portugal (Bjorndal *in litt.*) were recovered in Nicaragua. Recently, there is a demand for loggerhead and green turtle meat as bait in

lobster traps and the hook and line fishery for shark. Almost nothing is known about populations of loggerheads and leatherbacks in Caribbean waters of Nicaragua or their use of this habitat.

The status of green, hawksbill, and possibly loggerhead populations in the greater Caribbean, will depend, in part, on fishing activities of the Miskitu Indians. Offshore waters of Caribbean Nicaragua are home to a large number of sea turtles representing several species and nesting populations, as well as, the influx of animals from other areas of developmental habitat throughout the greater Caribbean. Thus, the legal, unregulated marine turtle fishery of Caribbean Nicaragua has the real possibility of severely impacting marine turtle populations throughout the greater Caribbean and needs to be assessed.

To date, no attempt has been made to evaluate the impact of the Nicaragua marine turtle fishery on turtle populations, although marine turtles have been harvested from these waters for at least 400 years and the area supports the largest remaining green turtle foraging population in the western Atlantic Ocean. Prior to the current study, only cursory information about the fishery was available and nothing was known about its current status. For these reasons, I focused my research on quantifying and describing 1) human patterns and use of the marine turtle harvest and 2) biological parameters of the animals harvested. From these data, I have conducted a preliminary evaluation of the impact of this fishery on marine turtle populations in the region using absolute and relative capture efforts and biological parameters of the harvested animals over time. The evaluation of the fishery is based on data collected over a relatively short period of time

(in relation to generational time of the resource), however, these results provide base-line information with which to compare subsequent years of harvest data. An evaluation of the impact of the fishery is necessary to provide the basis for developing management strategies to regulate the fishery.

This dissertation is organized into six chapters, including the current chapter. In Chapter 2, I describe and characterize the human patterns and use of the marine turtle harvest, including a description of fishery participants, harvest locations, harvest methods and their efficiency, and the human distribution of harvested animals. In Chapter 3, I quantify the number of harvested animals by species, size, and sex for the majority of the Caribbean coast of Nicaragua and analyze harvest rate and changes in size over time. In Chapter 4, I describe the reproductive cycle and status of a subset of harvested green turtles. In Chapter 5, I conduct a preliminary evaluation on the impact of the fishery based on historical information from the region, indices of current capture effort and demographics of harvested animals over time, and results from the literature on population modeling of long-lived organisms. In the final chapter, Chapter 6, I make recommendations for the development of a co-managed marine turtle fishery and provide management recommendations for the fishery based only on biological constraints of the species. Management options for the marine turtle fishery that can impinge on social, economic, and cultural aspects of the turtlers, turtle butchers, and coastal inhabitants will need to be discussed and agreed on among the turtlers, turtling-community representatives, and regional and central government officials. I also provide recommendations for future research.

CHAPTER 2
HUMAN USE PATTERNS

Introduction

Natural Resource Use

Throughout the world natural resources are harvested to meet dietary, medicinal, cultural, religious, and financial human needs and wants (e.g., Robinson and Redford 1991, 1994; Jorgenson 1993; Rose 1993, 1996; Bodmer 1994; Bodmer et al. 1994; Jenkins and Broad 1994; Bissonette and Drausman 1995; Jenkins 1995; Townsend 1995; Vincent 1996; Freese 1997). As a result, many plant and animal populations have been severely reduced or depleted. To mitigate negative impacts of human use or to aid in the recovery of depleted populations, it is necessary to manage human resource use.

Effective resource management cannot occur without a knowledge of human use patterns, such as, how and where resources are harvested, uses of the resources, and who are the beneficiaries. It is also important to quantify harvest effort, yield, and distribution of the harvest among resource consumers. These types of data can aid in the development and implementation of management schemes that mitigate restrictions imposed on resource users, meanwhile, improving compliance with regulations and the probability of long-term resource availability. In addition, monitoring changes in human use patterns, such as, harvest effort and rates, and areas of resource extraction, can

provide information about the sustainability of the harvest and the impacts of human use on resource populations. Thus, the identification and quantification of human use patterns are important for management schemes to be successful and can provide data critical in monitoring population trends in a resource.

Human Use of Sea Turtles

Historical use of sea turtles by humans has been well documented in chronicles of early travelers and by the scientific community because of strong interest in these animals (Hornell 1927; Ingle and Smith 1949; Parsons 1962; Dampier 1968; O'Donnell 1974; Frazier 1980; Bjorndal 1982). Marine turtles are exploited for their eggs, meat, shell, skin, and other products. Although much of the literature is descriptive, a few studies have quantified human use patterns of sea turtles, e.g., the use of olive ridley (*Lepidochelys olivacea*) eggs in Honduras (Lagueux 1989, 1991) and green (*Chelonia mydas*) and hawksbill (*Eretmochelys imbricata*) turtles in the Solomon Islands (Broderick pers. com.), Seychelles (Mortimer 1984), and Nicaragua (Nietschmann 1972, 1973, 1979a; Weiss 1975, 1976).

Marine turtles on the Caribbean coast of Nicaragua have been harvested by Amerindians since before the arrival of Europeans to the New World. These peoples, now known as the Miskitu Indians, have long been recognized for their turtling skills (Parsons 1962; Roberts 1965; Dampier 1968; Nietschmann 1972, 1973). Their turtle harvesting methods were described in the early 1800s by Roberts (1965), in the mid-1800s by Squier (1965) and Bell (1989) and in the early 1900s by Conzemius (1932).

Detailed accounts of natural resource use by two Miskitu turtling communities were reported by Nietschmann (1972, 1973) and Weiss (1975, 1976).

Subsequent to the studies by Nietschmann (1972, 1973, 1979a) and Weiss (1975, 1976) in the late 1960s and early 1970s, there have been several major events in eastern Nicaragua that have had a potential impact on human use patterns of marine turtles on this coast. Between 1969 and 1977, three marine turtle slaughter houses opened and closed; in 1977, Nicaragua became a signatory of the Convention of International Trade in Endangered Species (CITES; Hemley 1994); and most recently, in 1990, the country ended a 10-yr long civil war. As a result of these events, human use patterns described by Nietschmann (1972, 1973, 1979a) and Weiss (1975, 1976) are probably no longer indicative of current use patterns for the region. Their studies also lack a broader perspective of Miskitu Indian turtling practices because they focussed on resource use in a single community.

The current Miskitu and Rama Indian marine turtle fishery is a legal, uncontrolled harvest of green and hawksbill turtles. Although, in 1965, the Nicaragua government established a closed season to protect green turtles in their Caribbean waters for several months/yr (Nietschmann 1972, 1973; Rebel 1974; Weiss 1976; Bacon 1981; Peralta Williams 1991; Montenegro Jiménez 1992; pers. obs.) the closed season has been ineffective. The duration of the closed season has, apparently changed from two months, 15 May to 15 July, in the 1970s (Nietschmann 1972, 1973; Rebel 1974; Weiss 1976; Bacon 1981) to four months, 1 April to 31 July, in the 1980s and 1990s (Peralta Williams 1991; Montenegro Jiménez 1992 pers. obs). Although restrictions under the law are not

clear, the closed season has at different times varied in duration and included a total ban against the harvest of turtles, a ban against the commercialization of marine turtles, and a ban against the harvest of females (Nietschmann 1972, 1973; Weiss 1976; Montenegro Jiménez 1992). Several sources agree, however, that the law is not enforced, and some have argued that it is unenforceable (Nietschmann 1973; Peralta Williams 1991; D. Castro pers. com.). This is evident by the number of green turtles of both sexes landed during the closed season months (see Chapters 2 and 3). In addition, in 1997, enforcement of the closed season displaced the sale of green turtles from Puerto Cabezas to the Río Coco region of the country (D. Castro pers. com.), located on the border with Honduras. Thus, in 1997, the result of the closed season was that green turtles were distributed to non-traditional markets rather than reducing the harvest.

Human use patterns need to be determined to understand current impacts on turtle populations, monitor changes in human use patterns, and to improve our ability to successfully manage natural resource use. In this chapter, I characterize the following aspects of the turtle fishery: who fishes for marine turtles, how and where turtles are captured, and uses of marine turtles. In addition, I analyze the capture rate and human distribution of harvested turtles on temporal and spatial scales. These data can be used as a basis in the development of management recommendations for the marine turtle fishery. These data are not only important in a current evaluation of the fishery but also for establishing baseline levels with which future data can be compared. Identifying changes in capture per unit effort also provides a means to indirectly monitor turtle population trends and can be used as an indicator of overharvest and population decline.

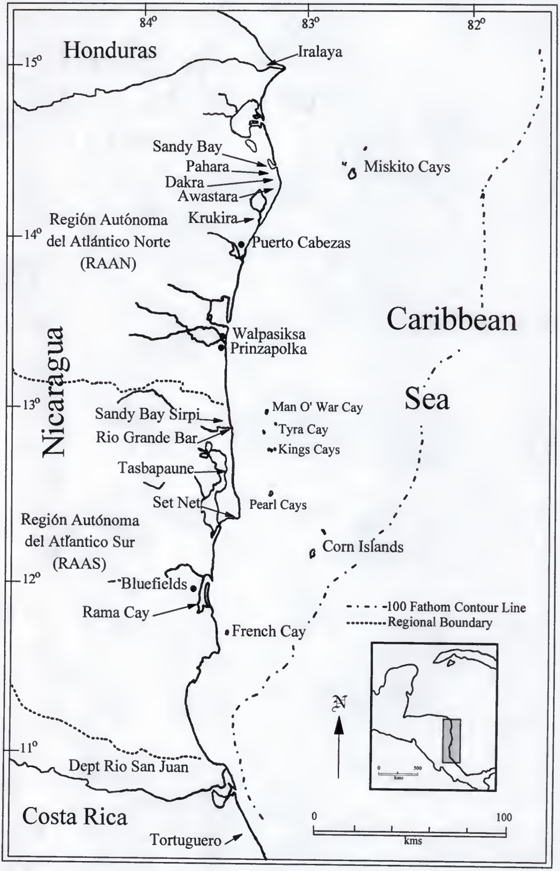
Methods

Study Site

Research was conducted on the Caribbean coast of Nicaragua (Figure 2.1). Politically, the eastern one-third of the country, including the inhabited Corn Islands and numerous offshore cays, is divided primarily into the Región Autónoma del Atlántico Norte (RAAN) and the Región Autónoma del Atlántico Sur (RAAS). Three coastal commercial centers are located in these regions: Puerto Cabezas in the RAAN, and Bluefields and Corn Island in the RAAS. Outside these commercial centers people on the coast reside in the following ethnically identified communities: Miskitu Indian, Miskitu Indian/Creole mix, Carib, and Rama Indian. Creoles are a racial and cultural mixture of African, European, and Amerindian traits (Hale and Gordon 1987). Caribs, or Garifunas as they are also known, are of African and Amerindian descent. They arrived in Nicaragua via Honduras as displaced slaves from the Caribbean island of St. Vincent (Hale and Gordon 1987).

On the Caribbean coast of Nicaragua, turtlers are Miskitu Indians, Creoles, or Rama Indians. In the RAAN, turtlers are Miskitu Indians. Most are bilingual, speaking Miskitu and Spanish, and many of the older inhabitants are trilingual, also speaking English. Miskitu is the language of everyday use in the RAAN and people identify themselves ethnically as Miskitu.

Figure 2.1. Caribbean coastline of Nicaragua with coastal communities and towns.


84°
83°
82°
15°
14°
13°
12°
11°
Honduras
Iralaya
Sandy Bay
Pahara
Dakra
Awastara
Krukira
Miskito Cays
Región Autónoma
del Atlántico Norte
(RAAN)
Puerto Cabezas
Caribbean
Sea
Walpasiksa
Prinzapolka
Nicaragua
Sandy Bay Sirpi
Rio Grande Bar
Man O' War Cay
Tyra Cay
Kings Cays
Tasbapaune
Set Net
Pearl Cays
Región Autónoma
del Atlantico Sur
(RAAS)
Corn Islands
Bluefields
Rama Cay
French Cay
100 Fathom Contour Line
Regional Boundary
N
Dept Rio San Juan
Costa Rica
Tortuguero
0
100
kms

In the RAAS, the ethnic identity of the turtling communities is Miskitu Indian, Miskitu Indian/Creole mix, or Rama Indian. In the Miskitu Indian/Creole mix communities, inhabitants are from a Miskitu ancestry but many have assimilated a Creole identity (Hale 1987; R. Carlos pers. com.). Creole English is spoken in the Miskitu Indian/Creole mix communities, particularly among the younger inhabitants. In one RAAS turtling community it was reported that many of the younger people no longer speak Miskitu (R. Carlos pers. com.). Rama Indian communities are located south of Bluefields. Residents from at least two of these communities harvest sea turtles.

The continental shelf of Nicaragua extends approximately 200 km at its widest point eastward from Cabo Gracias a Dios (near the Honduras/Nicaragua border) to approximately 20 km wide at its narrowest extension near the Costa Rican border. This extensive shelf provides Nicaragua with a vast area of productive marine ecosystems including mangrove and coral cays, underwater reefs, and seagrass pastures that supports many commercially and locally valuable resources, such as, shrimp, lobster, scale fish, and four species of marine turtles.

The marine turtle fishery occurs in Nicaragua's offshore Caribbean waters. The main turtling area in the RAAN is located offshore approximately 48 to 80 km. The Miskitu Indian/Creole mix communities located along the northern coast of the RAAS, turtle in areas located approximately 16 to 24 km offshore. Rama Indian turtling areas are located just offshore near French and Pigeon (located just north of French Cay) Cays. In addition, turtles can be captured just offshore in the RAAN and RAAS as they migrate to and from the nesting beach located at Tortuguero, Costa Rica (Figure 2.1).

Geographic and Temporal Distribution of Data Collection

In the RAAN, at the time of the study, five communities were known to fish for green turtles. These were Awastara, Dakra, Krukira, Pahra, and Sandy Bay (Figure 2.1). Beginning in April 1996 green turtles harvested by the community of Walpasiksa were also landed at Puerto Cabezas. Data on the turtle harvest were recorded in the communities of Awastara, Dakra, and Sandy Bay and when turtles were landed at the commercial center of Puerto Cabezas. Data on the harvest of sea turtles by the communities of Krukira, Pahra, and Walpasiksa were recorded only when turtles were landed in Puerto Cabezas because there were insufficient funds and the residents of Krukira were unwilling to participate in the study.

In the RAAS north of Bluefields, four communities were known to fish for green turtles, at the time of the study. These were Río Grande Bar, Sandy Bay Sirpi, Set Net, and Tasbapaune (Figure 2.1). Along the southern coast of the RAAS, south of Bluefields, at least two communities of Rama Indians also fish for turtles but were not included in the study due to financial constraints. Data on the turtle harvest in the RAAS were recorded in the four communities listed above. At the time of the study, no additional Caribbean Nicaragua communities were known to fish for marine turtles.

I trained local community residents to collect the data. In addition, I conducted informal interviews in communities and commercial centers during several short trips to Nicaragua in April and May 1992, November and December 1995, December 1996, and during a longer period of residence on the coast from November 1993 to February 1995. Data on who participates in the turtle fishery, types of boats used, and methods used to

capture turtles were collected opportunistically. In the indigenous communities, my key informants were community judges, secular and religious leaders, and turtlers; in commercial centers they were the turtle butchers and employees of each municipality.

At each of the eight turtling communities and commercial center, one or two residents of the site were employed as data collectors. Data collectors were selected based on their: 1) acceptance in this role by community members, 2) interest and availability in conducting the work, 3) ability to conduct the work, and 4) prior experience in collecting and recording data. Three of the data collectors employed in this study had worked previously with other researchers and had experience recording various types of data. I trained each data collector and supervised data collection. When a turtle boat arrived at any of the eight sites data collectors recorded the following: 1) turtlers community of residence, 2) date, 3) number of days turtling, 4) capture method used, 5) if nets were used, how many, 6) capture location, 7) number of animals of each species captured, and 8) where captured turtles were consumed or sold.

The period for which data are available from each site varies and is not always continuous (Table 2.1). Inconsistencies were due to the following factors: differing initiation dates of data collection, insufficient funds to continue employment of collectors, or unforseen circumstances (e.g. datasheets were stolen from the data collector or damaged by rain). Depending on the site, the number of months for which data were collected ranged from 29 mo for Awastara and Dakra to 64 mo in Sandy Bay Sirpi. The percentage of months for which data were collected within the data collection period ranged from 66.7% for Río Grande Bar to 100% for Set Net (Table 2.1).

Table 2.1. Summary according to site of the time period during which data collection occurred and the percent and number of months for which data were collected on the landing of marine turtles on the Caribbean coast of Nicaragua.

Site of Data Collection	Time Period (Duration of Period)	Percent of Months Data Collected (Number of Months)
Región Autónoma del Atlántico Norte		
Awastara	February 1994-April 1997 (39 months)	74.4 (29)
Dakra	February 1994-April 1997 (39 months)	74.4 (29)
Puerto Cabezas[a,b]	May 1991-April 1997 (72 months)	80.6 (58)
Sandy Bay[b]	May 1992-April 1997 (60 months)	83.3 (50)
Región Autónoma del Atlántico Sur		
Río Grande Bar[c]	April 1991-December 1996 (69 months)	66.7 (46)
Sandy Bay Sirpi[c]	January 1991-December 1996 (72 months)	88.9 (64)
Set Net	July 1994-December 1996 (30 months)	100 (30)
Tasbapaune	November 1993-December 1996 (38 months)	94.7 (36)

[a] Source of data for 1991: Cecil Clark, Puerto Cabezas, Nicaragua.

[b] Source of data for 1992 and 1993: Caribbean Conservation Corporation.

[c] Source of data for 1991-1993: Centro de Investigaciones y Documentación de la Costa Atlántica (CIDCA), Bluefields, Nicaragua.

Data Analysis

Because the turtling grounds in the RAAN and RAAS are separated by approximately 220 km of ocean, data were not combined for these two regions. Data were analyzed either at the regional or community levels. Mean green turtle "Net Capture Per Unit Effort" (N-CPUE) was calculated for each community and region. For each trip, the N-CPUE was calculated using the following parameters: 1) "Days" = number of days turtling, 2) "Nets" = number of nets used, 3) "Turtles" = number of green turtles captured, and 4) "Net-Days" were calculated as "Nets" $*$ "Days"; thus N-CPUE = "Turtles" $\div$ "Net-Days". Informal interviews with turtlers, indicated that turtle nets are approximately the same length, depth, and mesh size.

Tasbapaune was the only community that reported the capture of green turtles by both entanglement nets and harpoons. Thus, data collected in Tasbapaune were used to compare the relative effectiveness of nets and harpoons for capturing turtles. In order to make this comparison, I devised a more general index of CPUE (G-CPUE) for each trip using the following parameters: 1) "Days" = number of days turtling, 2) "Persons" = number of turtlers, 3) "Person-Days" = "Persons" $*$ "Days"; thus G-CPUE = "Turtles" $\div$ "Person-Days".

All statistical analyses were conducted using SAS software (SAS Institute, Inc. 1989). Univariate procedures were used to determine if distributions approximate normality and a t-test was used to test for equality of variances. When assumptions for parametric analyses were not met, non-parametric tests were used. Means $\pm$ 1 S.D. are presented.

Results

Turtlers and Living Conditions on the Turtling Grounds

Turtlers are male, usually older than 12 years of age, although younger boys can accompany turtlers and assist in preparing food, tending the fire, cooking, cleaning, and guarding the living quarters. Living conditions on the turtling grounds differ between the RAAN and RAAS. In the RAAS, fishers (including men from all fishing activities except for mechanized shrimp boats) have constructed semi-permanent structures on several of the numerous coralline cays. In the RAAN, however, offshore cays are covered with mangroves and are not habitable. In the RAAN, fishers eat and sleep on their boats or spend the night in *casitas* (shelters constructed over shallow water from mangrove poles, wood planks, palm thatch, and corrugated tin). On the boats, food is cooked over wood fires contained in a metal kettle and fishers sleep on the floor.

Transportation and Capture Methods

In the RAAN, turtlers use wood-planked sail boats, referred to locally as "dories". Dories are approximately 9 m long with a depth of 1 m from the gunwale to the boat floor and approximately 1.8 m at the widest point. They are constructed from hand-sawn planks of hardwood and powered by a main sail and jib. In the RAAS, turtlers that set nets use boats powered by 10-15 hp outboard motors to reach the turtling grounds, however, once they have arrived oars are used to set the nets and move between net-sets. Outboard motor boats used for turtling are approximately 9 m long, 0.6 m deep, and 0.6 m wide between the gunwales at the widest point. Turtle harpooners use "cayucos",

which are dug-out canoes powered by sail and paddles. Cayucos are approximately 4 m in length and 0.8 m deep.

In the RAAN, a fishing trip is usually focused on only one resource, however lobster divers also capture turtles, and harvest conch opportunistically. In the RAAS, a fishing trip is not resource specific. During a single trip the crew could set nets for turtles, dive for lobster, set lobster pots, and fish for shark. It was not possible to determine the number of active turtle boats per community in either the RAAN or RAAS because boats can be 1) used to harvest different resources from week to week, 2) under repair or construction, 3) rented out to a neighboring community, or 4) renamed once the boat is repaired or repainted.

In both the RAAN and RAAS, green turtles are captured primarily with entanglement nets. However, turtlers in Set Net and Tasbapaune (RAAS) continue to strike turtles with harpoons. Entanglement nets are constructed from No. 18 nylon twine and are from 21 to 27 m long and approximately 7.5 m deep, with a 40 to 47-cm bar mesh size. Discarded flotation material found on the beach is strung on the headline and pieces of coral, harvested from the reefs, are tied-in to the footline of the net to maintain it vertical in the water column. A larger piece of coral is used as an anchor. Nets are set during the day, over reefs or coral outcroppings, where green turtles are known to return after having foraged on grassbeds throughout the day. At night, when an animal rises to the surface to breath it entangles in the net. Rarely is more than one turtle captured in a net at a time. Turtles rarely die in the nets because they are able to rise to the surface while entangled in the net and breath throughout the night. In the early morning, nets and

any captured turtles are retrieved. Flippers are restrained by binding together each anterior flipper with a posterior flipper with twine passed through a slit made in the distal portion of each flipper. In the RAAN, animals are stored on their carapace (upside down) in the bottom of the dory until the end of the trip. In the RAAS, animals are stored in turtle crawls, or on their carapace on the cays for the duration of the fishing trip. Turtle crawls are pens enclosed by mangrove poles set vertically in shallow water. Animals are allowed to move freely within them.

Harpoons are constructed with a mahogany or palm wood shaft approximately 3 m in length. Inserted into one end of the shaft is a removable, triangular, 3-barbed point approximately 5.5 cm in length made from a metal file. The point is attached to a rope and at the other end of the rope is attached a wooden float. The shaft of the harpoon can also be attached to the rope. Two-man crews, a "striker" and a "captain", search for turtles early and late in the day. The striker poses on the bow of the boat ready to strike any turtle that surfaces within their range, approximately 12 m (D. Castro pers. com.). When a turtle is spotted the striker throws the harpoon, releasing the shaft and lodging the point in the carapace causing the point to detach from the wooden shaft. The captain quickly paddles the boat so that the striker can retrieve the float and shaft. The rope and turtle are hauled into the boat.

Although green turtles are the focus of the turtle fishery other species are sometimes captured. Hawksbill, loggerhead (*Caretta caretta*), and leatherback (*Dermochelys coriacea*) turtles are captured incidentally in nets set for green turtles. Hawksbills are also captured by harpoon, and by hand when diving for lobster.

Capture Effort

Spatial and temporal comparison of entanglement nets

From December 1995 to December 1996, RAAS turtlers captured more than twice as many green turtles/net-day as RAAN turtlers ($\bar{x}$ N-CPUE = 0.26 ± 0.17 in the RAAS and 0.12 ± 0.08 in the RAAN; Wilcoxon rank-sums, $P < 0.0001$). The RAAS turtlers spent fewer days turtling, used fewer nets, and captured more turtles/trip then RAAN turtlers (Table 2.2). By community, the highest $\bar{x}$ N-CPUE in the RAAN (0.14 ± 0.08 for Sandy Bay) is similar to the lowest $\bar{x}$ N-CPUE in the RAAS (0.13 ± 0.07 for Río Grande Bar) (Table 2.2).

Temporal change in mean monthly N-CPUE was analyzed for all sites combined in the RAAN and RAAS separately, and for the community of Sandy Bay Sirpi, RAAS. In the RAAN, mean monthly N-CPUE was calculated from December 1995 to April 1997 (17 mo) and in the RAAS, from December 1995 to December 1996 (13 mo). By region, there was no correlation between mean N-CPUE and month, and neither slope was significantly different from zero (RAAN, $P = 0.36$, $r = 0.03$; RAAS, $P = 0.85$, $r = -0.06$). For Sandy Bay Sirpi, mean monthly N-CPUE was calculated from January 1991 to December 1996 (data are available for 48 mo of the 72 mo period). There is a weak correlation between mean N-CPUE and month, however, the slope is not significantly different from zero ($P = 0.22$, $r = 0.37$).

Table 2.2. Comparison of green turtle, *Chelonia mydas*, capture effort using entanglement nets by community and combined for the Región Autónoma del Atlántico Norte (RAAN) and Región Autónoma del Atlántico Sur (RAAS), Nicaragua from December 1995 to December 1996. Mean ± 1 S.D. is followed by range and sample size. Means were calculated per trip.

Location	Turtles	Nets	Days Turtling	Net-Days[a]	N-CPUE[b]
RAAN					
Awastara	10.6 ± 6.0 1 - 27 120	17.8 ± 4.4 10 - 30 117	5.5 ± 2.4 1 - 14 120	99.7 ± 51.4 13 - 270 117	0.13 ± 0.09 0.02- 0.50 117
Dakra	5.3 ± 3.5 1 - 20 80	20.3 ± 3.4 15 - 36 80	3.9 ± 1.6 1 - 8 80	79.4 ± 37.6 27 - 200 80	0.07 ± 0.04 0.01 - 0.23 80
Sandy Bay	13.9 ± 6.7 2 - 40 124	25.9 ± 4.0 4 - 34 120	4.2 ± 1.4 1 - 8 124	108.8 ± 39.1 16 - 224 120	0.14 ± 0.08 0.02 - 0.63 120
Combined	10.6 ± 6.7 1 - 40 324	21.5 ± 5.4 4 - 36 317	4.6 ± 2.0 1 - 14 324	98.0 ± 45.0 13 - 270 317	0.12 ± 0.08 0.01 - 0.63 317
RAAS					
Río Grande Bar	28.4 ± 21.5 4 - 96 40	41.3 ± 13.9 7 - 58 40	5.7 ± 1.6 3 - 12 40	242.4 ± 116.8 21 - 660 40	0.13 ± 0.07 0.03 - 0.29 40
Sandy Bay Sirpi	9.2 ± 5.1 0 - 22 123	18.8 ± 4.5 8 - 30 119	3.1 ± 1.3 1 - 8 120	59.7 ± 31.5 12 - 200 118	0.19 ± 0.15 0 - 1.05 118
Set Net	5.2 ± 5.1 1 - 30 46	10.6 ± 2.7 5 - 16 44	2.3 ± 0.8 1 - 4 46	25.3 ± 12.6 9 - 60 44	0.22 ± 0.16 0.06 - 0.75 44
Tasbapaune	13.9 ± 3.9 3 - 24 167	15.1 ± 4.5 8 - 30 165	2.9 ± 1.0 1 - 6 167	43.8 ± 19.1 10 - 108 165	0.36 ± 0.13 0.13 - 0.91 165
Combined	12.8 ± 10.2 0 - 96 376	18.6 ± 10.3 5 - 58 368	3.2 ± 1.4 1 - 12 373	68.3 ± 76.0 9 - 660 367	0.26 ± 0.17 0 - 1.05 367

[a] Net-Days / Trip = (Number of nets / Trip) (Number of days turtling / Trip).
[b] Net Capture per Unit Effort = (Number of turtles / Trip) / (Net-Days / Trip); mean green turtle N-CPUE is significantly different between the RAAN and RAAS (Wilcoxon rank-sums, $P < 0.0001$).

Harpoon use

Set Net and Tasbapaune are the only two communities where harpoons were reportedly still in use. Between July 1994 and December 1996, Set Net reported the use of harpoons for only two turtling trips, therefore data on the capture effort of turtles with harpoons will be analyzed only for Tasbapaune. From December 1995 to December 1996 (13 mo), harpoons were used during 55 turtling trips. The mean G-CPUE was 2.7 ± 0.9 (Table 2.3).

Table 2.3. Comparison of the efficiency of harpoons and entanglement nets as methods for capturing green turtles, *Chelonia mydas*, by the community of Tasbapaune, Nicaragua from December 1995 to December 1996. Mean ± 1 S.D. is followed by range and sample size. Means were calculated per trip.

Capture Method	Turtles	Persons	Days Turtling	Person-Days[a]	G-CPUE[b]
Harpoons	5.6 ± 1.8 2 - 12 55	2.0 ± 0.0 2 - 2 55	1.0 ± 0.2 1 - 2 55	2.1 ± 0.4 2 - 4 55	2.7 ± 0.9 1.0 - 6.0 55
Nets	13.9 ± 3.9 3 - 24 167	3.0 ± 0.5 1 - 4 166	2.9 ± 1.0 1 - 6 167	8.6 ± 3.3 3 - 20 166	1.8 ± 0.8 0.7 - 6 166

[a] Person-Days / Trip = (Number of persons / Trip) (Number of days turtling / Trip).
[b] G-CPUE = (Turtles captured / Trip) / (Person-Days / Trip); mean green turtle G-CPUE is significantly different between the use of harpoons and nets (ANOVA, $P < 0.0001$).

Harpoons and entanglement nets compared

Since Tasbapaune turtlers did not use harpoons and nets during the same turtling trip, the effectiveness of netting and harpooning turtles can be compared statistically. From December 1995 to December 1996 (13 mo), the G-CPUE was significantly higher using harpoons (2.7 ± 0.9) than nets (1.8 ± 0.8; ANOVA, $F_{1,219} = 60.7$, $P < 0.0001$).

However, because netters spent nearly three times as many days turtling per trip than harpooners they captured more than twice as many turtles per trip (Table 2.3).

Capture Locations: Región Autónoma del Atlántico Norte (RAAN)

Between May 1992 and April 1997 (60 mo), a total of 66 capture locations for green turtles were recorded for the RAAN. Capture locations are places where turtlers claim to capture turtles with either nets or harpoons. The approximate surface area of capture locations range from 0.1 km^2 to 6.5 km^2. Hawksbills were captured at 23 of the 66 (34.8%) capture locations between December 1993 and April 1997 (41 mo), and loggerheads at 27 of the 66 (40.9%) capture locations between September 1994 and April 1997 (32 mo).

A total of 41 green turtle capture locations were reported by turtlers from Awastara, Dakra, Sandy Bay, and Puerto Cabezas for the same period (February 1994 to January 1995, and December 1995 to April 1997, 29 mo). Hawksbills were captured at 20 (48.8%) and loggerheads at 25 (61.0%) of these 41 capture locations. The greatest percentage of green turtles (40.5%, n = 4,702), hawksbills (26.9%, n = 21), and loggerheads (39.9%, n = 252) were captured at Witties. An additional 27.6% (n = 174) of loggerhead captures occurred at Leimarka. Each of the remaining capture locations produced less than 11% of the turtles captured for the three species. Only trips with no more than one capture location reported were included in these analyses. See Appendix A for a detailed compilation of RAAN capture locations for each species.

The RAAN communities overlapped in their use of capture locations. Twenty-one of 41 (51.2%) capture locations were used by more than 1 of the 6 communities (including data collected from the communities of Krukira, Pahra, and Walpasiksa when they landed their turtles at Puerto Cabezas) and 6 (14.6%) capture locations were used by 4 or 5 communities. None of the capture locations were used by all six communities. Of the 19 (46.3%) capture locations used by only 1 community, 11 (26.8%) locations were used exclusively by Sandy Bay, 6 (14.6%) exclusively by Awastara, 2 (4.9%) exclusively by Dakra, and 1 location was used only by Walpasiksa (Appendix A).

Capture Locations: Región Autónoma del Atlántico Sur (RAAS)

Between January 1991 and December 1996 (72 mo), a total of 77 capture locations for green turtles were recorded for the RAAS. From August 1994 to December 1996 (29 mo), hawksbills were captured at 30 (39.0%) and loggerheads at 24 (31.2%) of the 77 capture locations. However, a large proportion of animals for each species was captured from a small proportion of the capture locations.

There was nearly complete partitioning of the turtling grounds between the four RAAS communities. Turtlers from Río Grande Bar and Sandy Bay Sirpi shared three capture locations and turtlers from Set Net and Tasbapaune shared only one capture location. Río Grande Bar and Sandy Bay Sirpi did not share any capture locations with Set Net or Tasbapaune. Because there was almost no overlap in the use of the 77 capture locations among the RAAS turtling communities results for each community are

presented separately. See Appendix B for a detailed compilation of RAAS capture location data for each community and species.

Río Grande Bar. For 46 mo, between April 1991 and December 1996, 24 green turtle capture locations were recorded. The majority of green turtles (53.8%, n = 3,027) were captured at three locations (Half-way, Vietnam, and Karmutra Banks). The remaining 21 capture locations each yielded less than 8% of the green turtles captured by Río Grande Bar turtlers. Hawksbills were captured at eight (33.3%) and loggerheads at six (25.0%) of the 24 capture locations reported by Río Grande Bar. Half-way Bank yielded the greatest percentage of hawksbills (25.0%, n = 3), whereas, De Tronco yielded the greatest percentage of loggerheads (40.0%, n = 4).

Sandy Bay Sirpi. For 64 mo, between January 1991 and December 1996, 34 green turtle capture locations were recorded. Three locations (Wainwin, South Schooner, and Half-Way Bank) accounted for 34.6% (n = 1,709) of the green turtles captured. The remaining 31 capture locations each yielded less than 8% of the green turtles captured by Sandy Bay Sirpi turtlers. Hawksbills were captured at 9 (26.5%) and loggerheads at 8 (23.5%) of the 34 capture locations reported by Sandy Bay Sirpi. Three locations, Hawksbill Bank, Halfway Bank, and Lousiksa accounted for 72.3% (n = 26) of the hawksbills captured and two locations, Family Shoal and Diamond Spot, accounted for 64.3% (n = 18) of the loggerheads captured.

Set Net. During the 30 mo period from July 1994 to December 1996, 7 green turtle capture locations were recorded. One location, Fowlshit Bank, accounted for 64.2% (n = 488) of the green turtles captured. Hawksbills were captured at 5 (71.4%) and

loggerheads at 3 (42.9%) of the 7 capture locations. Long Reef accounted for 50.0% (n = 9) of the hawksbills captured and together with Fowlshit Bank accounted for 95% (n = 19) of the loggerheads captured.

Tasbapaune. For 36 mo, between November 1993 and December 1996, 14 green turtle capture locations were recorded. Four locations (Haulover, Rivas, Buscan, and Middle Banks) accounted for 71.6% (n = 4,392) of the green turtles captured. The remaining 10 locations each yielded less than 9% of the green turtles captured by Tasbapaune turtlers. Hawksbills were captured at 9 (64.3%) and loggerheads at 7 (50.0%) of the 14 capture locations. One location, Haulover Bank, accounted for 30.8% (n = 20) of the hawksbills captured. The capture of loggerheads was more evenly distributed, with 3 of the 7 capture locations each accounting for between 23% and 32.5% of the total loggerhead captures.

Use and Human Distribution of Harvested Turtles

Green turtles

Green turtle meat is traditionally consumed by humans. It is either used for subsistence by the turtlers, their families and friends, or sold in local and regional markets. More recently, the meat is also used as bait in lobster pots and for shark fishing. Turtles are transported to markets by the turtlers or purchased from the turtlers at sea or in their communities and transported to other markets by the buyer.

Región Autónoma del Atlántico Norte (RAAN). Green turtles were distributed among at least 11 local and regional markets between November 1993 and April 1997 (42

mo). These markets are located as far north as Iralaya, Honduras and as far south as Bluefields and Corn Island (Figure 2.1). The relative amounts of meat consumed within and outside the turtlers community of residence differed between the three turtling communities. For the period from February 1994 to January 1995, and from December 1995 to April 1997 (29 mo), for which monthly data collection occurred for all three communities, Awastara turtlers sold the majority of their green turtles (85.2%) outside their community of residence, whereas, the majority of turtles harvested by Sandy Bay (84.1%) and Dakra (60.9%) were consumed in the turtlers community of residence (Table 2.4). Less than 5.5% of the turtles harvested by each community were consumed during turtling trips (Table 2.4).

Of the green turtles sold to other markets, 82.0% (n = 5,895) were sold in the commercial center of Puerto Cabezas. The remaining 18.0% (n = 1,291) were sold among at least nine other markets (Table 2.4). A small amount of turtles sold by Awastara (1.5%) and Dakra (2.1%) were sold to the turtling community of Sandy Bay.

Región Autónoma del Atlántico Sur (RAAS). Green turtles were distributed among at least 15 local markets between January 1991 and December 1996 (72 mo). These markets are located between Sandy Bay Sirpi and Corn Island (Figure 2.1). For the period from January to December 1996 (12 mo), for which monthly data collection occurred for all four communities, Río Grande Bar (89.0%), Sandy Bay Sirpi (60.5%), and Set Net (61.1%) turtlers sold the majority of their green turtles outside their community of residence. Tasbapaune, however, consumed the majority (71.5%) of the

Table 2.4. Number and (percent) of green turtles, *Chelonia mydas*, by location of consumption for three Región Autónoma del Atlántico Norte (RAAN) turtling communities from February 1994 to January 1995 and December 1995 to April 1997 (29 mo). All markets are located within Nicaragua unless otherwise indicated.

	RAAN TURTLING COMMUNITIES			
Location of Consumption	Awastara	Dakra	Sandy Bay	Total
Community	774 (10.7)	1,022 (60.9)	3,351 (84.1)	5,147 (39.9)
Other Markets				
Bihmuna	0 (0)	63 (3.8)	5 (0.1)	68 (0.5)
Bluefields	174 (2.4)	0 (0)	0 (0)	174 (1.4)
Cabo Gracias a Dios	0 (0)	71 (4.2)	23 (0.6)	94 (0.7)
Corn Island	382 (5.3)	20 (1.2)	15 (0.4)	417 (3.2)
Iralaya, Honduras	0 (0)	41 (2.4)	2 (0.1)	43 (0.3)
Koom	0 (0)	0 (0)	10 (0.3)	10 (0.1)
Puerto Cabezas	5,215 (72.2)	364 (21.7)	316 (7.9)	5,895 (45.7)
Río Coco	0 (0)	0 (0)	8 (0.2)	8 (0.1)
Sandy Bay	107 (1.5)	35 (2.1)	0 (0)	142 (1.1)
Sold on cays[a]	202 (2.8)	0 (0)	48 (1.2)	250 (1.9)
Unknown	72 (1.0)	13 (0.8)	0 (0)	85 (0.7)
Turtling Trip	297 (4.1)	48 (2.9)	207 (5.2)	552 (4.3)
Not Consumed (died)	3 (0.04)	0 (0)	0 (0)	3 (0.02)
Total	7,226	1,677	3,985	12,888

[a] Sold to Colombian, Cuban, Honduran, and Nicaraguan commercial fishing boats for human consumption, and lobster trap and shark bait.

turtles harvested by their turtlers. Less than 6.5% of the turtles harvested by each community were consumed during turtling trips (Table 2.5).

Of the green turtles sold to other markets, 67.4% (n = 1,571) were sold in the commercial center of Bluefields. The remaining 32.6% (n = 760) were sold among nine other markets (Table 2.5). Río Grande Bar sold 15.2% (n = 144) of the turtles it sold to other markets to the turtling community of Sandy Bay Sirpi.

Temporal change in human distribution of harvested turtles. The human distribution of harvested green turtles was compared between years for five turtling communities using a Chi-square test. Data were compared between 1994 and 1996 for Awastara, Dakra, Sandy Bay, and Sandy Bay Sirpi; and between 1995 and 1996 for Tasbapaune. There was a significant difference in the human distribution of harvested green turtles from one year to the next for each turtling community (Chi-square test, $P < 0.001$, Table 2.6). From 1994 to 1996, for the three RAAN communities (Awastara, Dakra, and Sandy Bay), there was an increase in the proportion of animals consumed in the community, whereas, for the two RAAS communities (Sandy Bay Sirpi and Tasbapaune) there was an increase in the proportion of animals sold outside the community. For four of the five communities, the percent of green turtles consumed during the turtling trip increased from 1994 to 1996, however, this category still represents a small proportion ($\leq 6.6\%$) of the fate of harvested green turtles (Table 2.6).

Hawksbill turtles

Hawksbills are harvested for their scutes. The meat can be consumed by the turtler and his family, given to others for consumption, or discarded. From January 1991

Table 2.5. Number and (percent) of green turtles, *Chelonia mydas*, by location of consumption for the Región Autónoma del Atlántico Sur (RAAS) turtling communities from January 1996 to December 1996 (12 mo). All markets are located within Nicaragua. RGB = Río Grande Bar, SBS = Sandy Bay Sirpi, SN = Set Net, TA = Tasbapaune.

	RAAS TURTLING COMMUNITIES				
Location of Consumption	RGB	SBS	SN	TA	Total
Community	97 (9.1)	418 (37.1)	68 (32.7)	1,762 (71.5)	2,345 (48.2)
Other Markets					
Bluefields	603 (56.6)	572 (50.6)	5 (2.4)	391 (15.9)	1,571 (32.3)
Cocabilla	0 (0)	0 (0)	17 (8.2)	0 (0)	17 (0.3)
Corn Island	80 (7.5)	30 (2.7)	0 (0)	84 (3.4)	194 (4.0)
Haulover	0 (0)	0 (0)	3 (1.4)	0 (0)	3 (0.1)
Kukra Hill	0 (0)	10 (0.9)	0 (0)	0 (0)	10 (0.2)
Marshall Point	0 (0)	0 (0)	1 (0.5)	0 (0)	1 (0.02)
Orinoco	0 (0)	0 (0)	0 (0)	57 (2.3)	57 (1.2)
Pearl Lagoon	0 (0)	0 (0)	101 (48.6)	9 (0.4)	110 (2.3)
Sandy Bay Sirpi	144 (13.5)	0 (0)	0 (0)	0 (0)	144 (3.0)
Sold on cays[a]	121 (11.4)	0 (0)	0 (0)	2 (0.1)	123 (2.5)
Unknown	0 (0)	71 (6.3)	0 (0)	30 (1.2)	101 (2.1)
Turtling Trip	20 (1.9)	26 (2.3)	13 (6.3)	130 (5.3)	189 (3.9)
Not Consumed (died)	0 (0)	0 (0)	0 (0)	1 (0.04)	1 (0.02)
Total	1,065	1,127	208	2,466	4,866

[a] Sold to Honduran and Nicaraguan commercial fishing boats.

Table 2.6. Human distribution of harvested green turtles, *Chelonia mydas*, compared between years for five turtling communities. Percent of year's total is followed by (number) of green turtles. Years within a community were compared using a Chi-square test.

		DISTRIBUTION OF HARVESTED GREEN TURTLES			
Community	Year	Consumed in Community	Sold Outside Community	Consumed on Trip	P
Awastara	1994	4.5 (132)	94.7 (2,771)	0.8 (23)	< 0.001
	1996	14.5 (440)	78.8 (2,385)	6.6 (201)	
Dakra	1994	45.6 (453)	53.6 (532)	0.8 (8)	< 0.001
	1996	89.4 (330)	4.9 (18)	5.7 (21)	
Sandy Bay	1994	78.6 (1,279)	18.2 (296)	3.3 (53)	< 0.001
	1996	88.9 (1,428)	4.8 (77)	6.3 (101)	
Sandy Bay Sirpi	1994	48.7 (384)	51.3 (404)	0 (0)	< 0.001
	1996	37.1 (418)	60.6 (683)	2.3 (26)	
Tasbapaune	1995	79.9 (127)	9.4 (15)	10.7 (17)	< 0.001
	1996	71.4 (1,759)	23.3 (573)	5.3 (130)	

through 1996, at least 272 hawksbills were captured in the RAAN and RAAS (see Chapter 3 and Appendix F). Although most captured hawksbills are killed, key informants reported that sometimes scutes are removed from live animals by leaving them in the hot sun, holding the carapace over a fire, or peeling off the scutes with the hot blade of a knife. The live, scuteless animal is then released. Coastal inhabitants reported

seeing marked animals with regenerated scutes, however, the regeneration of scutes has not been verified nor have mortality rates from this practice been quantified. Scutes are dried and stored until a buyer is found. Local artisans make various types of jewelry from hawksbill shell which can be found for sale throughout the country including in commercial centers along the coast, at the international and national airports, and at tourist markets in Managua.

Loggerhead turtles

Loggerhead turtles are either released from the nets unconscious, killed and discarded, or harvested for shark and lobster trap bait; the meat is not consumed because of its strong flavor. Turtlers kill or club loggerheads unconscious to facilitate removal from the nets, and to avoid the risk of being bitten. Mortality is high even among those animals that are discarded while still alive because clubbed loggerheads are usually discarded prior to regaining consciousness and most probably drown. From January 1994 through 1996 , at least 825 loggerheads were reported captured (see Chapter 3 and Appendix F).

Discussion

Capture Methods

The capture methods described by Nietschmann (1972, 1973, 1974) and Weiss (1975, 1976) are relatively unchanged except that outboard motors are now used by RAAS net-turtlers, but not by harpoon-turtlers. Outboard motors are a relatively recent addition and are a result of negotiations to end the civil war in the mid-1980s (D. Castro

pers. com.). The RAAS fishers were provided with loans from the central government. In the RAAN, however, turtlers continue to use sailing dories. Outboard motors are used in the RAAS to travel between the mainland and turtling grounds, but not to aid in the capture of turtles. Their use decreases travel time to and from the turtling grounds, but because motors are not used to set or retrieve nets the use of outboard motors in the RAAS does not account for the greater net capture rate.

Entanglement nets is the principal method in use today to capture turtles. Nets were introduced to the Miskitu Indians by Cayman Island turtlers sometime before 1915 (Conzemius 1932). Prior to their introduction, Miskitu Indians used harpoons. Today, only two of the Miskitu and Miskitu Indian/Creole mix communities (Tasbapaune and Set Net) continue to use harpoons. In the late 1960s, the use of both nets and harpoons by Tasbapaune turtlers was reported but no mention was made as to the relative prevalence of either method (Nietschmann 1972, 1973). At least as early as 1971, harpoons were no longer used by Sandy Bay Sirpi turtlers (Weiss 1975) and this pattern continues today.

The use of nets provides an advantage for neophyte turtlers, however, the harvesting of coral to anchor-down the nets could be detrimental to the environment. Because net-setters work in crews of 3 to 6 men, an inexperienced turtler has the advantage of working with more experienced turtlers and will be successful at capturing turtles and thus profit immediately. The opportunity to learn from more experienced turtlers while sharing in the profits derived from their success could explain the prominent use of nets today. One of the drawbacks to the use of nets is the indiscriminate

harvest of coral for use as weights on the footline and to anchor the net. Studies are needed to quantify the amount of coral harvested and to evaluate the extent of damage to the reef.

Historically, harpoons were the sole method of capture, however, today, they are seldom used. Their decreased use could reflect the need to be more skillful in order to be successful. Harpooners probably spend more time practicing and need more patience to become proficient, and the technique necessitates good communication and cooperation between the striker and captain. Nevertheless, the increased investment in time needed to be a proficient harpooner can be worthwhile in the long-run because harpoons are a more efficient method of capturing turtles, at least during this study. However, Nietschmann (1973) reported nets were more efficient, although the basis for this conclusion is not given.

Capture Efficiency

On average, RAAN turtlers spend more days turtling and use more nets/trip but capture fewer green turtles than turtlers in the RAAS. The difference between the regions in the mean number of days spent turtling/trip is probably due to the distance of the turtling grounds to the communities and the mode of travel. The RAAN turtlers travel more than twice the distance than RAAS turtlers to reach most of their turtling grounds and they travel by sail-powered dories compared to the use of motorboats by RAAS turtlers. Because of the greater time invested by RAAN turtlers to reach the turtling grounds it's not surprising that they spend more days turtling/trip.

The higher net capture rate of green turtles in the RAAS could reflect either higher turtle density or better turtling skills by RAAS turtlers. Both regions of the country, however, have been turtling for several hundred years, have been exposed to similar outside influences and technology, and currently use the same netting technique. Therefore the observed difference in capture rates is more likely explained by a difference in green turtle abundance due to physiognomic and biotic habitat variation, turtle migration patterns, or exploitation rates during the recent past. Mortimer (1981) found that stomach contents of green turtles differed among several capture locations along the Nicaragua coast suggesting that floristic composition varied among locations. Other parameters, such as substrate type, current direction and flow rate, water depth and clarity could also influence capture efficiency. In addition, storms can cause local perturbations affecting habitat and subsequently sea turtle populations. Green turtles feed selectively on young blades of turtle grass, *Thalassia testudinum*, thereby, decreasing the proportion of lignin and increasing the proportion of protein consumed (Bjorndal 1980a). In 1988, Hurricane Joan struck the coast of Nicaragua near Bluefields (Roth 1992). Although no studies were conducted to identify changes that occurred to the offshore underwater habitats in the RAAS as a result of the storm, its possible that the disturbance caused by the hurricane stimulated new growth among the seagrass beds resulting in higher green turtle population densities.

Secondly, funneling of turtles into a relatively narrow pathway as they migrate south towards the nesting beach at Tortuguero, Costa Rica could increase turtle density seasonally as turtles from the RAAN and farther north migrate south through the RAAS.

Turtles reportedly often use a longshore route when passing through Nicaraguan waters during breeding migrations, traveling along the coast in nearshore waters (Carr 1954). Therefore, in Nicaragua, when turtles use these longshore routes turtlers set their nets over the shallow, mud flats, referred to as a "mudset", located within 3 km of shore between Prinzapolka and Set Net point (see Figure 2.1, Carr 1954; Nietschmann 1973; Mortimer 1976, 1981).

A third possibility is that exploitation levels in the two regions have differed during the recent past, thus affecting current resource abundance. In the RAAN, during the civil war, turtlers were only allowed to make trips to the offshore cays that originated from Puerto Cabezas. For the period 1985 to 1990 (during the Sandinista/Contra war), 16,700 green turtles ($\bar{x}$ = 2,783 turtles/yr, S.D. = 681, range = 1,619 - 3,375, n = 6) were harvested from the RAAN and landed in Puerto Cabezas (Montenegro Jiménez 1992). For the RAAS, harvest levels of marine turtles during the war are not available, however, turtling activities were more severely decreased or suspended because of the large military presence and battles fought near the communities. Thus, the higher capture rate of turtles in the RAAS could be due to a higher density of turtles because of reduced harvest pressure during the war compared to the RAAN.

Mean, monthly, N-CPUE in the RAAN, RAAS, and for Sandy Bay Sirpi does not indicate, at this time, that the abundance of turtles within the size range of animals harvested has declined during the study period. The trend in the N-CPUE for all three analyses indicates there has been no change in the number of green turtles captured/net-day. However, analyses were conducted over very short time periods (from 13 to 72 mo)

and could be too short to detect a change in the abundance of the foraging population. A problem with using CPUE to estimate stock abundance is that fishers go where the fish or turtles are and CPUE can remain high until the stock is seriously depleted, a situation described as hyperstability by Hilborn and Walters (1992).

The N-CPUE for Sandy Bay Sirpi has increased 850% since the study conducted by Weiss (1975) in 1972/1973, assuming net length, depth, and mesh size have remained constant. Based on data provided by Weiss (1975), I calculated a mean of 0.02 turtles/net-day captured for Sandy Bay Sirpi turtlers for a one-year period beginning in mid-1972 compared to a mean of 0.19 ± 0.15 turtles/net-day calculated for the period December 1995 to December 1996 (this study). The low N-CPUE realized in 1972/1973 could reflect declines in the turtle population resulting from the estimated 6,000 - 10,000 animals harvested annually from the coast between 1969 and 1973. Current harvest levels for the coast are similar to those reported for the late 1960s and early 1970s. Although this harvest level has not been continuous since the 1970s, Sandy Bay Sirpi harvest levels have been relatively constant since at least 1991 (see Chapter 3). In 1990, the country ended a decade-long civil war during which the exploitation of natural resources by both nationals and foreigners was greatly reduced (Nietschmann 1995). The harvest of turtles in the RAAS may not be as intense and the number of turtlers may have decreased since the 1970s because today many men also fish for lobster and shark. Whereas, in 1972, Weiss (1976) reported the only other major source of income in Sandy Bay Sirpi was as temporary wage laborers on shrimp and fishing boats.

Capture Locations

The 141 capture locations for marine turtles range in surface area from 0.1 km^2 to 6.5 km^2. Because of their vast size, each location has within it numerous sites where nets are set. Marine turtle capture sites are dynamic, with new sites identified and known sites abandoned due to low productivity. Turtlers identify potential new capture sites by looking for submerged rocks on sunny, clear days. Nets are set and if turtles are captured the site is given a name. According to the turtlers, each turtle uses several different "sleeping rocks". Because sites within capture locations can be temporarily unproductive, turtlers do not set their nets at the same site every day. Capture sites can also become unproductive for long periods, e.g., in December 1995, P. Hills, a Río Grande Bar turtler, reported 13 unproductive capture sites. No information is available as to how long a capture site can remain unproductive.

Partitioning of turtling grounds among the communities differs between the RAAN and RAAS. In the RAAN, communities overlap extensively in their use of turtle capture locations, whereas, in the RAAS, there is almost complete partitioning by the communities. The lack of partitioning in the RAAN is probably because the distance to the turtling grounds is approximately the same regardless from which community a trip originates. Geographically, the RAAN communities are located relatively close to each other but the turtling grounds are distant. In the RAAS, turtling grounds are located offshore of each community decreasing the need to overlap on the turtling grounds or travel farther than necessary. This partitioning of capture locations by community in the RAAS was recognized as early as 1972 (Weiss 1975).

In the RAAN, managing the use of capture locations should be discussed at a regional level with representation from each of the turtling communities. For example, one of 35 capture locations in the RAAN (Witties) provided a large percentage of the green (40.5%), hawksbill (26.9%), and loggerhead (39.9%) turtles captured. Therefore, if reduction in the use of this one capture location could be agreed upon by all RAAN turtlers a decrease in the capture rate of all three species could be realized.

The fact that some communities overlap in their use of capture locations could facilitate enforcement. For example, compliance with regulations on the use of capture locations could be enhanced by intercommunity surveillance. Because the proportion of each community's harvest is not the same from each capture location, however, some form of compensation might have to be devised so that any hardships can be more equitably distributed among the communities.

In contrast, the establishment of regulations governing the use of capture locations in the RAAS would best be approached on a community by community basis. Although selection and agreement of regulations on the use of capture locations could be easier to establish on a community by community basis, compliance and self-surveillance could be more difficult to achieve. Since turtles are not confined in their movements by community partitioning of capture locations, but probably move among the RAAS capture locations and possibly between RAAN and RAAS foraging areas it will also be important for the RAAS communities to develop a regional management plan, as well as, inter-regional agreements for the use of this shared resource.

Human Distribution of Harvested Turtles

Approximately 50% of the green turtles captured are sold outside the turtlers communities of residence, both in the RAAN (55.7%) and the RAAS (48.0%). The percent of turtles distributed to outside markets varies greatly from one community to the next, from a low of 10.8% for Sandy Bay to a high of 89.0% for Río Grande Bar. The decision to sell the harvest outside the community probably depends on the size of the community, purchasing power of the community, and the turtlers need to obtain money.

Dakra, Sandy Bay, and Tasbapaune, communities with the largest number of inhabitants, consumed more than 60% of the green turtles harvested by their turtlers. In 1994, the populations of Dakra and Sandy Bay were approximately 1,400 and 4,000 inhabitants, respectively (Comisión Nacional Interinstitucional 1995). In 1996, Tasbapaune had approximately 3,200 inhabitants (R. Carlos pers. com.). In addition, the capacity of Sandy Bay inhabitants to consume turtles was apparently greater than the number of turtles harvested because they purchased 1.5% of Awastara's and 2.1% of Dakra's turtle harvests.

The ability of a community to purchase turtles is dependent on the communities overall purchasing power. The higher the median income in a community the more money available to purchase goods and services, including turtle meat. Beginning in 1995, Sandy Bay and Dakra received assistance from private companies to enter into the lobster trap fishery (D. Castro pers. com.). This has probably increased the median income in each community. The increase in the percent of turtles consumed in Sandy Bay and Dakra from 1994 to 1996 is possibly a result of this additional source of income.

Río Grande Bar, Awastara, Set Net, and Sandy Bay Sirpi consumed the smallest percent of their harvests in their communities, 9.1%, 10.7%, 32.7%, and 37.1%, respectively. Of these communities, Río Grande Bar and Set Net had the fewest inhabitants, with approximately 150 and 200 inhabitants, respectively in 1996 (L. Churnside pers. com.; F. Thomas pers. com.). In contrast, Awastara and Sandy Bay Sirpi are large communities with approximately 1,200 inhabitants in 1994 (Comisión Nacional Interinstitucional 1995) and 1,100 inhabitants in 1995 (E. Smith pers. com.), respectively. Compared to Sandy Bay and Dakra, two other RAAN turtling communities, Awastara has fewer sources of income and although they increased their community consumption of turtle meat from 4.5% in 1994 to 14.5% in 1996 they still consume a relatively small proportion of their turtle harvest.

The human distribution pattern of harvested turtles by Sandy Bay Sirpi is less easily explained. Although Sandy Bay Sirpi is one of the largest of the turtling communities it sold over 71% of its harvested turtles outside the community. However, it also purchased 13.5% of Río Grande Bar's turtles. For Sandy Bay Sirpi, the incentive to sell a large majority of its green turtle harvest outside the community might be motivated by a higher price/lb of meat sold in the Bluefields market compared to a lower price for live turtles purchased from another turtling community. Turtle meat is sold in the communities for approximately 1/3 to 1/2 the price/lb of meat sold in the commercial centers (Lagueux unpubl. data).

The proportion of the harvest sold outside the turtlers community of residence during this study is lower than the proportion sold to the turtle processing plants in the

early 1970s, for the two communities for which data are available. In 1971, Tasbapaune sold 66% of its harvest to the plants (Nietschmann 1972, 1973) compared to 23.3% of the harvest sold outside the community today. For a one-yr period, from 1972 to 1973, Sandy Bay Sirpi sold 81.4% of its harvest to the plants (Weiss 1976) compared to 60.5% of the harvest sold outside the community today.

These data probably underestimate the amount of trade occurring on the foraging ground themselves. Because of the expanse of the area and the number of boats fishing on Nicaragua's continental shelf there is a large potential for trade in sea turtles with other boats. Informants report that on the cays, turtles are purchased to feed the crews of fishing and lobster boats from San Andres Colombia, Cuba, Honduras, and Nicaragua, and also purchased for transport to the Cayman Islands and San Andres Colombia to be sold in their markets. More effort is needed to determine, for each species, the number of animals purchased for bait, purchased by fishing boats for food, and transported to other countries.

Conclusions

The patterns of human use of sea turtles in Nicaragua today are not much different than during the past several hundred years of resource use on this coast. Since prior to European contact, Miskitu Indians have harvested marine turtles from their coastal waters for personal use, to trade for other goods, and to sell for income with which to purchase other goods and services. Although the current harvest of marine turtles is no longer to supply international markets, the local and regional (within Nicaragua) demand for a

source of inexpensive protein and tortoiseshell could be just as devastating to the marine turtle foraging populations. As Frazier (1980) has noted, the issue is not whether the exploitation of a resource is “good” or “bad” but whether or not the resource can sustain harvest levels and patterns of resource use.

Data presented in this chapter are needed for the establishment of a management plan for the Miskitu marine turtle fishery. These data are important to our understanding of current human use patterns of marine turtles and are beneficial in making management decisions. In addition, these data provide a basis with which to monitor changes in the fishery which may result from management actions or due to external forces.

CHAPTER 3
HARVEST RATES AND DEMOGRAPHICS OF MARINE TURTLES

Introduction

Throughout history the harvest of sea turtles and their eggs have occurred virtually wherever people and turtles coincide. Although the eggs of all species have been harvested, animals of the seven species have been exploited to varying degrees. Most harvests of marine turtles have targeted green turtles (*Chelonia mydas)* and hawksbills (*Eretmochelys imbricata)* (Hornell 1927; Ingle and Smith 1949; Carr 1954; Parsons 1956, 1962, 1972; Hirth and Carr 1970; Nietschmann 1972, 1973; Rebel 1974; Frazier 1975, 1979; Cato et al. 1978; Bjorndal 1982; Dodd 1982; King 1982; Milliken and Tokunaga 1987; Mortimer 1984; Meylan 1997a). Harvests of the olive ridley (*Lepidochelys olivacea)* (Carr 1967, 1972, 1979; Pritchard 1979; Frazier 1981), Kemp's ridley (*Lepidochelys kempi)* (Pritchard and Márquez 1973), loggerhead (*Caretta caretta)* (Brongersman 1982), and leatherback (*Dermochelys coriacea)* (Starbird and Suarez 1994; Suarez and Starbird 1995) have either occurred for a shorter duration or have been more localized. The endemic flatback turtle (*Natator depressus)* has played a limited role in marine turtle harvests in Australia (Bustard 1972).

In Nicaragua, marine turtles have been harvested from beaches and offshore waters of the Caribbean coast by indigenous coastal inhabitants and foreigners for over

400 years (Parsons 1962; Roberts 1965; Dampier 1968; Nietschmann 1973; Montenegro Jiménez 1992; Lagueux 1993). Four of the world's seven extant marine turtle species are found offshore on Nicaragua's vast continental shelf -- greens, hawksbills, loggerheads, and leatherbacks. Nicaragua's coastal area provides foraging and developmental habitat for the largest green turtle foraging population in the Atlantic Ocean (Carr et al. 1978). Juvenile and adult turtles immigrate to Nicaragua's coastal waters from throughout the greater Caribbean. Green turtles also use these waters as a migratory pathway for travel to and from the nesting beach at Tortuguero, Costa Rica (Carr 1954). Juvenile and adult hawksbill turtles can be found foraging among offshore coral reefs and seagrass beds (Nietschmann 1973, 1981; Lagueux pers. obs.). The hawksbill is the only species that has been confirmed to nest on mainland beaches and offshore cays of Caribbean Nicaragua (Nietschmann 1973, 1981). Almost nothing is known about the use of Nicaragua's beaches and offshore waters by loggerheads and leatherbacks. Unconfirmed reports, however, indicate that green, loggerhead, and leatherback turtles nest infrequently on Nicaragua's mainland beaches (Bacon 1975; Carr et al. 1982; this study).

In the Nicaragua fishery, the green turtle is the principal species targeted. Hawksbill turtles are harvested opportunistically whenever they are encountered. Loggerhead and leatherback turtles, although not targeted in the marine turtle fishery, are captured incidentally in nets set for green turtles.

Historically, only rough estimates have been made on the magnitude of the Nicaragua marine turtle fishery (Lewis 1940; Ingle and Smith 1949; Carr 1954; Parsons 1962; Nietschmann 1973; Weiss 1975) and demographics of harvested animals have

never been reported. Because Nicaragua's marine turtle fishery is legal and not clandestine, collecting data on the fishery and harvested animals is more easily accomplished. Data on the number of animals captured by species, and size and sex of harvested animals are needed to evaluate the impact of the fishery on marine turtle populations that occur within Nicaraguan waters, as well as, the greater Caribbean region. In addition, the sex ratio and size distribution of captured green turtles are probably indicative of the foraging population because there is no evidence that net capture methods are biased towards either sex or size, within the size range of animals captured in the fishery. These data are also important in regulating and monitoring resource use.

In this chapter, I quantify the number of animals harvested by species per year and describe the size and sex of harvested animals. The magnitude of the harvest for each species is analyzed to determine seasonal and long-term trends. The size and sex ratio of harvested animals by species are determined to provide baseline information on the segment of each population impacted by the fishery, and to compare the demographics of harvested green turtles between regions of the coast. The size distribution of harvested female green turtles is compared to the size distribution of nesting females to determine the proportion of harvested animals that are smaller than reproductive size. A temporal analysis is conducted to determine if changes have occurred in the mean size of harvested animals.

Methods

Data Collection

The turtling communities and commercial center where data were collected are described in Chapter 2. The initiation date for the collection of harvest data on each turtle species differed. Partial data on the harvest of green turtles from both the Región Autónoma del Atlántico Norte (RAAN) and the Región Autónoma del Atlántico Sur (RAAS) have been available at least since 1991 and are included in this study. Data collection on the harvest of hawksbills was initiated January 1991 in the RAAS and December 1993 in the RAAN. Data collection on the capture of loggerheads and leatherbacks was initiated January 1994 in the RAAS and September 1994 in the RAAN. For all species, data collected through April 1997 in the RAAN and December 1996 in the RAAS are included.

In 1991, Cecil Clark, head of the marine turtle butchers cooperative recorded landings of turtles at Puerto Cabezas. From 1991 to 1993, the collection of harvest data in the communities of Río Grande Bar and Sandy Bay Sirpi was conducted by the Centro de Investigaciones y Documentación de la Costa Atlántica (CIDCA). Beginning in April 1992 in the RAAN and in November 1993 in the RAAS, I trained local data collectors. Supervision of data collection has been conducted by Denis Castro, my Miskitu Indian counterpart, and me. Selection and employment of data collectors are described in Chapter 2.

Types of Harvest Data Collected

Harvest and demographic data on marine turtles were recorded when a boat returned from a turtling trip. For each turtling trip, the following data were recorded: 1) turtlers community of residence, 2) date trip terminated, and 3) total number of turtles of each species captured. For as many individual turtles as possible, the following data were also recorded: 1) species, 2) plastron length (PL), and 3) sex. Plastron length was measured along the midline from the anterior junction of the skin and intergular scute to the posterior termination of the plastron midline with a 150-cm flexible tape measure. Although plastron length is not the preferred measurement taken among sea turtle biologists it was the most practical because animals are transported and stored upside down. Sex was based on the examination of external sex characteristics, i.e., tail length and the size and shape of the anterior flipper claws.

During a preliminary study conducted from May 1992 to March 1993 in Puerto Cabezas and from May 1992 to December 1993 in Sandy Bay, the following turtle measurements were recorded: 1) minimum (notch-to-notch) curved carapace length (CLN), 2) minimum (notch-to-notch) straight carapace length (SLN), 3) body mass (WT), and 4) sex. A 150-cm flexible tape measure was used for all curved measurements and a 127-cm tree caliper was used for all straight measurements. Minimum carapace lengths were measured along the midline from the anterior edge of the nuchal scute to the posterior termination of the midline to the nearest 0.1 cm. Body mass was determined to the nearest 2.5 lb with a 500-lb spring scale and converted to kilograms.

Calculation of Harvest Rates

The annual harvest rates of marine turtles are minimum estimates because harvest data were not collected from every Miskitu Indian and Miskitu/Creole turtling community nor from any of the Rama Indian communities (see Chapter 2). In addition, not all turtles captured were necessarily reported to the data collectors and none of the turtles captured incidental to other fisheries were reported. Monthly harvest rates for each data collection site were calculated based on the numbers of turtles actually recorded and estimated to have been landed. For a variety of reasons, however, data were not collected during every month of the study period at all data collection sites (see Methods in Chapter 2). For those months for which data were not available, I estimated the monthly harvest rate based on known monthly harvest rates for each site, for each year. In 1995, data were collected for only two to three months in Awastara, Dakra, and Sandy Bay. Thus, I estimated the monthly harvest rates for each data collection site based on 12 months of known harvest rates, 6 months prior and 6 months post the period of missing data.

Care was taken not to include the same harvested animal more than once in the totals. Because, in the RAAN, turtle boats dock at more than one data collection site it was possible for animals to be recorded twice. To avoid overestimating the harvest rate, I excluded from the total, animals recorded at one site but sent to another site in which a data collector was employed. However, if turtles were sent to a site where no data collector was employed then these animals were included in the total.

Turtle Morphometrics

A series of 11 body measurements were used to attempt to characterize morphological differences between male and female turtles foraging in Nicaragua. Because sea turtle biologists record any of eight different carapace lengths (Pritchard et al. 1983), it is difficult to make comparisons among studies. Therefore, regression equations were developed to predict the other measurements from minimum curved carapace (CLN). For green turtles, measurements were taken from a stratified random sample of animals landed at Puerto Cabezas between November 1993 and January 1995. Each month I attempted to measure a minimum of 40 harvested animals, 10 from each of the following four categories: "small" males, "large" males, "small" females, and "large" females. The cut-off between "small" and "large" was based on the size of the smallest nesting females at Tortuguero, Costa Rica in 1988 (Caribbean Conservation Corporation unpubl. data). Animals smaller than 89.8 cm minimum straight carapace length were categorized as "small". The sex of animals was verified when they were butchered (see Chapter 4). All hawksbills were measured whenever possible. No loggerheads or leatherbacks were measured because they are not landed at Puerto Cabezas. Any measurement that might have been affected by deformities or mutilations were excluded from the analyses.

The 11 body measurements I recorded were the following: 1) minimum (notch-to-notch) curved carapace length (CLN), 2) minimum (notch-to-notch) straight carapace length (SLN), 3) maximum (tip-to-tip) curved carapace length (CLT), 4) maximum (tip-to-tip) straight carapace length (SLT), 5) plastron length (PL), 6) plastron to vent length

(VENT), 7) tail length (TAIL), 8) length of each anterior flipper claw (CLEN), 9) minimum basal diameter of each anterior flipper claw, 10) maximum basal diameter of each anterior flipper claw, and 11) body mass (WT). Maximum carapace lengths were measured from the most anterior to the most posterior projections of the carapace. Vent and tail lengths were measured from the posterior termination of the plastron midline to the center of the vent (VENT) and to the tip of the straightened tail (TAIL). Length of each anterior flipper claw was measured from the junction of the skin and claw to the tip of the claw along the outer curvature and the mean CLEN/turtle ($\bar{x}$CLEN) was calculated from right and left measurements. Minimum and maximum basal diameters of each anterior flipper claw were measured with a dial caliper to the nearest 0.05 mm and used to calculate the basal area of each anterior flipper claw. Basal area was calculated using the formula for the area of an ellipsoid:

$$\mathrm{A} = \pi ab$$

where π is 3.14, a is one-half the shortest diameter, and b is one-half the longest diameter. Mean basal area of anterior claw/turtle ($\bar{x}$CBASE) was calculated from right and left measurements. All other measurement are as previously described (see Types of Harvested Data Collected in Methods). A 127-cm tree caliper was used for straight carapace measurements. Unless otherwise stated, measurements were made with a 150-cm flexible tape measure.

Accuracy of Sex Identification of Green Turtles

The ability of the Puerto Cabezas data collector to sex animals based on external characteristics was evaluated to determine the accuracy of the data obtained. He first sexed each animal externally, and I noted whether or not external sex characteristics were obvious (as described above). Sex was then verified based on gonadal examination of the butchered animals. Turtles were grouped into 2.5-cm size classes based on plastron length in order to identify the minimum plastron length at which an animal can be sexed using external characteristics.

Size and Life Stage Distribution of Harvested Green Turtles

Two data sets were analyzed to identify the life stage of green turtles impacted in the fishery. One data set is from the harvested animals landed at Puerto Cabezas during the preliminary study between May 1992 and March 1993. Only female carapace lengths recorded during the preliminary study were compared to nesting females at Tortuguero, Costa Rica. Data from the preliminary study were used because the data collectors ability to accurately sex animals based on external characteristics was found to be highly reliable (see Results on Sex ratio of harvested green turtles). The second data set is from the harvested animals measured and sexed between February 1994 and December 1996 at eight data collection sites where turtles were landed.

Using the preliminary data set, minimum straight carapace lengths (SLN) of harvested female green turtles landed at Puerto Cabezas, Nicaragua were compared to the maximum straight carapace lengths (SLT) of reproductively mature females measured on

the nesting beach at Tortuguero, Costa Rica (Caribbean Conservation Corporation unpubl data). Predicted SLN measurements were calculated from measured SLT of Tortuguero turtles. A regression equation to predict SLN was calculated from carapace lengths of animals I measured in Puerto Cabezas (see Methods on Turtle Morphometrics).

Using data collected at eight sites, the number of harvested male and female green turtles that were reproductively immature and mature were estimated for the RAAN and RAAS because smaller animals cannot be reliably sexed based on external characteristics (see Results on Sex ratio of harvested green turtles). For males, reproductive maturity was based on the presence of sperm, although this does not indicate that animals have reproduced (see Chapter 4). For females, reproductive maturity was based on the presence of corpora lutea in the ovaries (see Chapter 4) and the minimum size of females at the Tortuguero, Costa Rica rookery. Minimum plastron lengths of "mature" males and females were used as a cut-off between mature and immature animals. Estimates for the number of harvested males and females of reproductively immature and mature status were calculated by multiplying the number of harvested animals that were below and above the cut-off sizes for each sex by the proportion of harvested animals of each sex calculated for each region (see Results on Sex ratio of harvested green turtles).

Statistical Analysis

Seasonal and temporal harvest trends

Pearson correlation coefficient was used to determine if there was a relationship between the monthly number of green turtles harvested in the RAAN and monthly

rainfall recorded at Puerto Cabezas. Daily rainfall was recorded by either an assistant or me.

Analysis of variance was used to compare the RAAN mean monthly harvest rate of green turtles during this study with the mean monthly harvest rate calculated by Montenegro Jiménez (1992) from 1985 to 1990. Although data from Montenegro Jiménez (1992) were collected from landings of turtles only at Puerto Cabezas, her data represent the total harvest of animals in the RAAN at that time because fishing activities were restricted during the Nicaragua civil-war. For the RAAS, time series analysis was used to examine the trend in the monthly harvest of green turtles. Time series analysis also was used to examine the trend in the monthly recorded harvest of hawksbills and capture of loggerheads for all data collection sites combined. Trend data were examined for the possibility of autocorrelated residuals. A regression model with autoregressive errors was used when autocorrelation among error terms was significant at $P \leq 0.05$; however, when they were not, a simple linear regression model was used.

Turtle morphometrics

Pearson correlation coefficient was used to determine the relationship between body measurements and carapace length. Analysis of covariance was used to determine if males and females differed in their relationship between CLN and other body measurements. Dependent variables were log transformed when the assumptions of regression analysis were not met. Regression equations are reported separately when intercepts and slopes were significantly different between males and females, otherwise data were pooled and regression equations were recalculated.

Evaluating sex ratios

McNemar's test of dependent samples was used to analyze the accuracy of the Puerto Cabezas data collector to determine sex of animals based on external characteristics (Zar 1996). A Chi-square test was used to determine if the sex ratio of harvested green turtles in the RAAN and RAAS differed. A normal approximation to a binomial distribution (One-sample Proportion test) was used to determine if the sex ratio of harvested animals differed from a one-to-one ratio.

All statistical analyses were conducted using SAS software (SAS Institute, Inc. 1989). Univariate procedures were used to determine if distributions approximate normality and a t-test was used to test for equality of variances. When assumptions for parametric analyses were not met, non-parametric tests were used. Means ± 1 S.D. are presented.

Results

Green Turtles

Harvest levels

From 1991 to 1993, the estimated annual harvest ranged from 6,169 to 9,440 green turtles based on extrapolations of data collected at three or four sites, depending on the year (Figure 3.1). From 1994 to 1995, the estimated annual harvest ranged between 9,413 to 11,077 green turtles based on extrapolations of data collected at eight collection sites. In 1996, the minimum annual harvest was 10,166 green turtles based on

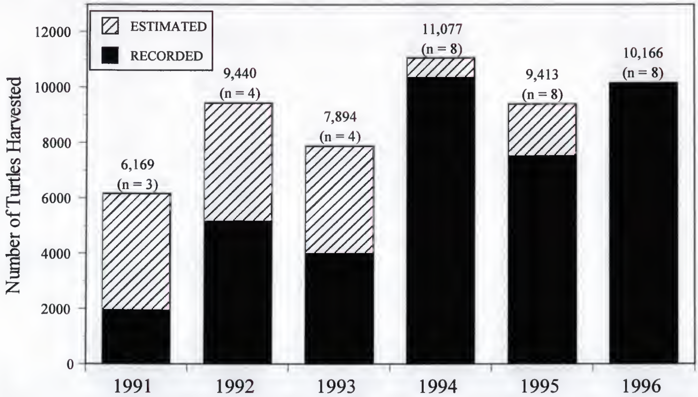


Figure 3.1. Minimum (i.e., recorded) and estimated (i.e., extrapolated) annual harvest of green turtles, *Chelonia mydas*, landed on the Caribbean coast of Nicaragua per year from 1991 to 1996. Sample size for each year refers to the number of sites where data were collected. See text for an explanation of how estimated data were calculated and see Appendix C for a list of sites represented in each year. Data were, in part, provided by C. Clark for 1991, Centro de Investigaciones y Documentación de la Costa Atlántica for 1991-1993, and Caribbean Conservation Corporation for 1992-1993.

data recorded at the same eight sites (Figure 3.1). Detailed harvest levels for each community are compiled in Appendix C.

Seasonal and temporal trends of harvest levels

Mean monthly rainfall from April 1994 to May 1995 was 221.6 mm ± 176.1 (range = 13.7 - 547.0 mm, n = 14). There was no correlation between monthly rainfall recorded in Puerto Cabezas and the number of green turtles harvested in the RAAN (P = 0.87, r = - 0.05).

In the RAAN, from 1985 to 1990 (during the Sandinista/Contra war), 16,700 green turtles ($\bar{x}$ = 245.6 turtles/mo ± 18.8, range = 0 - 739 turtles/mo, n = 68) were reported harvested (Montenegro Jiménez 1992). From February 1994 to January 1995 and from December 1995 to April 1997 (post Sandinista/Contra war), 14,017 green turtles ($\bar{x}$ = 483.3 turtles/mo ± 142.7, range = 131 - 745 turtles/mo, n = 29) were harvested in the RAAN (Figure 3.2). There was a significant increase in the number of green turtles harvested/mo from the Sandinista/Contra war to the post-war period (ANOVA, $F_{1,95} = 49.98$, $P < 0.0001$).

In the RAAS, from August 1994 to December 1996, 10,019 green turtles ($\bar{x}$ = 345.5 turtles/mo ± 171.8, range = 47 - 718 turtles/mo, n = 29) were harvested by four communities. Although the harvest rate has not changed significantly (P = 0.29, based on a regression model of autoregressive errors) there has been an increase of 58.7 turtles/yr during this 2.5-yr period (Y = 269.2 + 4.9X, where X is the number of months elapsed; Figure 3.3). Data for one RAAS community, Sandy Bay Sirpi, were available for a six-yr period which allowed the evaluation of harvest trends for a longer time period. From

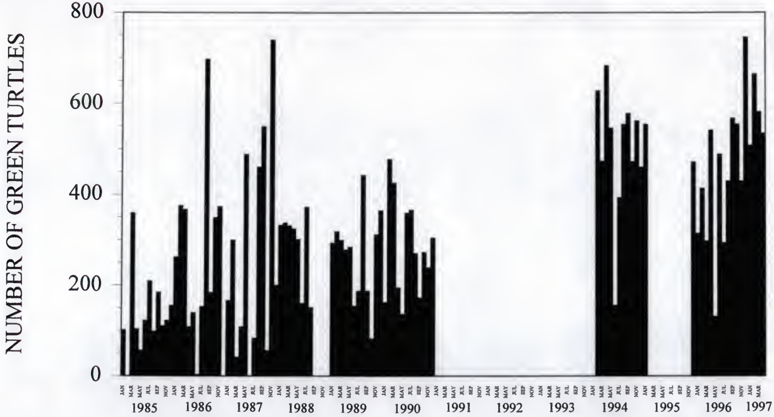


Figure 3.2. Number of green turtles, *Chelonia mydas*, harvested in the Región Autónoma del Atlántico Norte (RAAN), Nicaragua from January 1985 to April 1997 (149 mo). No data are available for months that appear as zeros. There was a significant increase in the mean number of turtles harvested/mo between the periods 1985 - 1990 and February 1994 - April 1997 (ANOVA, $F_{1,95} = 49.98$, $P < 0.0001$). Data for 1985 to 1990 taken from Montenegro Jiménez (1992).

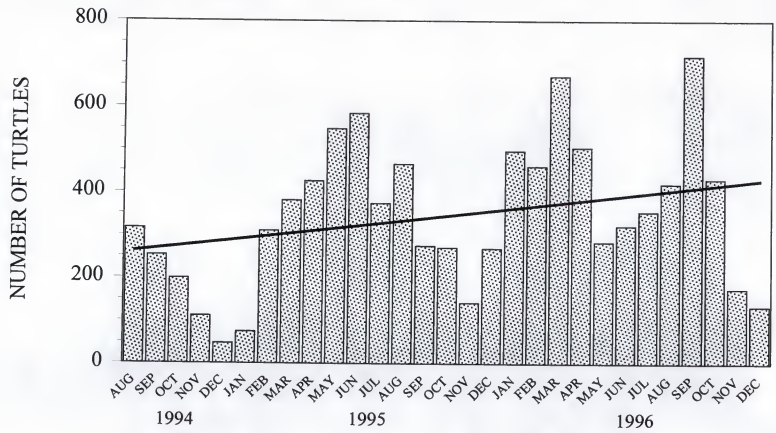


Figure 3.3. Number of green turtles, *Chelonia mydas*, reported harvested in the Región Autónoma del Atlántico Sur (RAAS), Nicaragua from August 1994 to December 1996 (29 months). There was no significant change in the number of turtles harvested/mo. The line is based on a least squares regression (turtles/mo = 269.2 + 4.9X, where X is the number of months elapsed; P = 0.29).

January 1991 to December 1996 (72-mo period), 5,036 green turtles ($\bar{x}$ = 85.4 turtles/mo ± 52.3, range = 3 - 237 turtles/mo, n = 59) were harvested. Based on a simple regression model the harvest rate of green turtles by Sandy Bay Sirpi turtlers has remained essentially constant at a decrease of 1.1 turtles/yr ($P = 0.78$) during this 6-yr period (Figure 3.4).

Morphometric parameters of harvested turtles

A total of 634 turtles were measured. For animals where sex was confirmed by dissection, minimum (notch-to-notch) curved carapace lengths (CLN) for females ranged from 75.0 - 113.2 cm (n = 276) and for males it ranged from 72.2 - 108.6 cm CLN (n = 282). Pearson correlation coefficients are high ($r > 0.89$) among the various carapace lengths (i.e., CLN, minimum straight carapace (SLN), maximum curved carapace (CLT), maximum straight carapace (SLT)); plastron (PL); and body mass (WT) for sexes combined or separate. Correlation coefficients for each of plastron-to-vent length (VENT), tail length (TAIL), and mean claw basal area ($\bar{x}$CBASE); regressed with each of CLN, SLN, CLT, SLT, PL, and WT are higher when sexes are separated ($r > 0.69$) than when sexes are combined ($r < 0.52$). Mean claw length ($\bar{x}$CLEN) is more highly correlated with all but one (PL, $r = 0.66$) of the other eight body measurements for males ($r > 0.73$) than for females ($r < 0.67$). Between 23% and 55% more of the variation in $\bar{x}$CLEN of males than females can be attributed to other body measurements. Coefficients are high among measurements of sexually dimorphic external characters ($\bar{x}$CBASE:VENT or TAIL) for males and sexes combined ($r \geq 0.92$) and lower for females (r = between 0.72 and 0.76). Plastron and VENT are highly correlated with

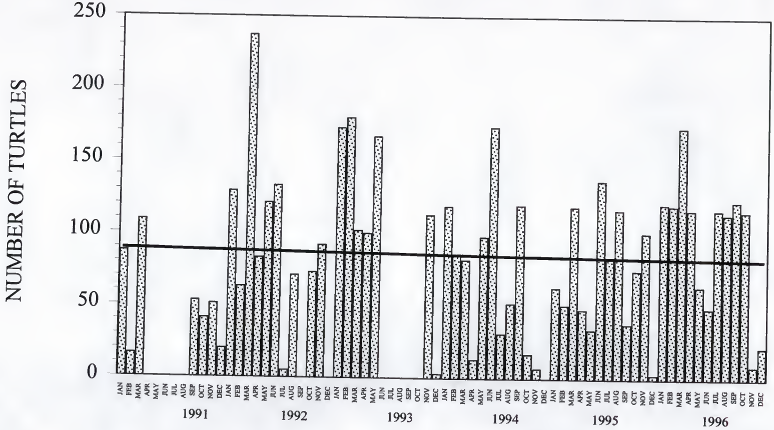


Figure 3.4. Number of green turtles, *Chelonia mydas*, reported harvested at Sandy Bay Sirpi, Nicaragua from January 1991 to December 1996 (72 months). There was no significant change in the number of turtles harvested/mo. The line is based on a least squares regression (turtles/mo = 89.0 - 0.092X, where X is the number of months elapsed; P = 0.78). No data are available for months that appear as zeros. Data for 1991 - 1993 provided by Centro de Investigaciones y Documentación de la Costa Atlántica.

TAIL for sexes combined or separate ($r > 0.97$). Correlation coefficients for all ten body measurements are shown in Appendix D.

There was no significant difference between males and females for the relationship between CLN and each of SLT, log PL, and log WT; therefore, data were pooled. There was, however, a significant difference between males and females for the relationship between CLN and each of CLT, SLN, log VENT, log TAIL, log $\bar{x}$CBASE, and log $\bar{x}$CLEN; therefore, regression equations were calculated separately by sex. Although analysis of covariance showed a significant difference between males and females for CLN:CLT and CLN:SLN, there is no apparent sexual dimorphism as indicated by the very high coefficients for the sexes combined and separate ($r = 0.99$ for both). Regression equations and statistical results are summarized in Appendix E.

Scatter plots of the 10 body measurements (each plotted against the other for a total of 45 pairs) were visually evaluated to determine if any pairs of measurements provide a means to distinguish between males and females. For 24 of the 45 pairs of measurements, divergence between males and females is obvious. However, not all pairs were equally divergent, nor was there complete divergence between males and females for any pair, particularly among the smaller sizes of animals. The greatest divergence between the sexes was observed in the plot of TAIL:WT, although TAIL plotted with CLN, SLN, CLT, SLT, PL (body lengths) were also highly divergent. Using the plot of TAIL:WT measurements, I developed a key to distinguish between males and females (Table 3.1). Although TAIL and WT were more easily distinguished on the scatterplot

Table 3.1. Keys to distinguish between male and female green turtles, *Chelonia mydas*, using tail length with either body mass or minimum curved carapace length (CLN). Scatterplots of tail length with body mass, and tail length with carapace length were used to distinguish between the sexes. For some measurements, however, sexes were indistinguishable (IND). Keys are based on confirmed males (n = 278) and females (n = 275). Cut-off measurements to distinguish between the sexes are approximate. Tail length = distance from posterior termination of the plastron midline to the tip of the straightened tail. CLN = distance along the midline from the anterior edge of the nuchal scute to the posterior termination of the midline. All animals measured were landed at Puerto Cabezas, Nicaragua between November 1993 and January 1995.

Tail Length (cm)	Body Mass (kg)			Carapace Length (cm)		
	♂♂	♀♀	IND	♂♂	♀♀	IND
<13		> 41	< 41		> 74	< 74
13-15		> 60	< 60	< 75	> 87	75 - 87
15-17	< 46	> 64	46-64	< 78	> 87	78 - 87
17-19	< 54	> 72	54-72	< 81	> 88	81 - 88
19-21	< 58	> 80	58-80	< 83	> 93	83 - 93
21-23	< 72	> 86	72-86	< 88	> 93	88 - 93
23-25	< 86	> 102	86-102	< 94	> 100	94 - 100
25-27	< 80	> 98	80-98	< 94	> 99	94 - 99
27-29	< 92	> 100	92-100	< 93	> 96	93 -96
29-31	< 90	> 90		< 96	> 96	
31-33	< 90	> 90		< 97	> 97	
33-35	< 110	> 110		< 100	> 100	
35-37	< 110	> 110		< 100	> 100	
>37	all males			all males		

between the sexes, I also developed a key based on TAIL:CLN because it is it is often difficult to weigh animals in the field (Table 3.1).

Sex ratio of harvested turtles

Between March 1994 and February 1995, the sex of 94.6% of 570 animals (size range = 72.2 - 113.2 cm minimum curve carapace length, CLN) were correctly identified by the Puerto Cabezas data collector using external characteristics. There was no difference in the misidentification of males or females (McNemar's test, $\chi^2 = 0.29$, df = 1, $P > 0.10$). In addition, I recorded whether or not the development of male external sex characteristics (i.e., the length of the tail and the size and shape of the anterior flipper claws) was obvious for 511 animals ranging in plastron (PL) length from 55.2 to 90.2 cm (CLN range = 72.2 - 113.2 cm). Based on the percent of animals, grouped by 2.5-cm size classes, with obvious external sex characteristics and the percent error of the data collector, I decided that the accuracy of sexing animals with a PL length ≥ 70.0 cm, based on external sex characteristics, was acceptable (Table 3.2).

For turtles with PL lengths ≥ 70.0 cm, there was a significant difference between the sex ratio of turtles harvested in the RAAN and RAAS (Chi-square test, $\chi^2 = 59.1$, df = 1, $P < 0.001$). In the RAAN, the sex ratio of males to females was 1:1.7 which differs significantly from a 1:1 ratio (One-sample Proportion test, $Z = 12.7$, $P < 0.0001$). In the RAAS, the sex ratio of males to females was 1:1.1 which also differs significantly from a 1:1 ratio (One-sample Proportion test, $Z = 3.1$, $P < 0.001$), however, this is probably not

Table 3.2. Accuracy of sex identification based on external characteristics with confirmation based on gonadal examination of harvested green turtles, *Chelonia mydas*, landed at Puerto Cabezas, Nicaragua from November 1993 to January 1995. The bold line at 70.0 cm is the minimum plastron length at which the sex of animals based on external characteristics was deemed acceptable. External characteristics are: the length of the tail in relation to the size of the animal and the size and shape of the anterior flipper claws.

Plastron Length (cm)	Sex Identification Based on External Characteristics		Number of Animals with Obvious External Sex Characteristics (%)		External Sex Identification Confirmed Based on Gonads		Percent Error
	Female	Male	Yes	No	Female	Male	
55.0-57.4	1	1	0 (0)	2 (100)	0	2	50.0
57.5-59.9	2	1	0 (0)	3 (100)	0	3	66.7
60.0-62.4	7	7	3 (21.4)	11 (78.6)	6	8	7.1
62.5-64.9	24	12	12 (33.3)	24 (66.7)	21	15	8.3
65.0-67.4	19	24	27 (62.8)	16 (37.2)	19	24	0
67.5-69.9	35	42	60 (77.9)	17 (22.1)	38	39	3.9
70.0-72.4	33	63	90 (93.8)	6 (6.3)	36	60	3.1
72.5-74.9	28	54	80 (97.6)	2 (2.4)	28	54	0
75.0-77.4	44	30	72 (97.3)	2 (2.7)	45	29	1.4
77.5-79.9	31	17	48 (100)	0 (0)	31	17	0
80.0-82.4	8	4	12 (100)	0 (0)	8	4	0
82.5-84.9	16	0	16 (100)	0 (0)	16	0	0
85.0-87.4	4	0	4 (100)	0 (0)	4	0	0
87.5-89.9	3	0	3 (100)	0 (0)	3	0	0
90.0-92.4	1	0	1 (100)	0 (0)	1	0	0
Totals	256	255	428 (83.8)	83 (16.2)	256	255	2.7

biologically significant. The odds of capturing a female ≥ 70.0 cm PL length in the RAAN are 1.6 times greater than capturing a female in the RAAS.

Size and stage distribution of harvested turtles

There was a significant difference in the size of green turtles harvested in the RAAN and RAAS (Wilcoxon Rank Sums, $Z = 34.6$, $P < 0.0001$). In the RAAN, mean PL length was 69.7 cm ± 6.8 (range = 38.0 - 96.4 cm, n = 7,209) and in the RAAS, it was 63.3 cm ± 11.5 (range = 24.4 - 95.4 cm, n = 8,613; Figure 3.5).

The size of females harvested in Nicaragua was compared to the size of nesting females at Tortuguero, Costa Rica to identify the life history stage of females harvested in the fishery. From April 1992 to March 1993, mean minimum straight carapace length (SLN) of females harvested in Nicaragua was 83.2 cm ± 7.5 (range = 64.2 - 113.8 cm, n = 899) and the mode was the interval 77.0 cm to 79.9 cm (Figure 3.6). In 1988, mean predicted carapace length (SLN) of nesting Tortuguero females was 98.2 cm ± 4.2 (range = 88.3 - 112.9 cm, n = 120). Although carapace measurements taken for the two studies differed, the Pearson correlation coefficient between them was high ($r = 0.99$, $P < 0.0001$). Therefore, I predicted SLN from straight maximum carapace length (SLT) measurements for Tortuguero females based on the equation $SLN = -1.358 + 1.002(SLT)$. There was a significant difference in mean carapace length (SLN) of harvested Nicaragua female green turtles and the mean predicted SLN of nesting Tortuguero females (Wilcoxon Rank Sums, $Z = 15.6$, $P < 0.0001$; Figure 3.6). Between May 1992 and March 1993, 78.2% of the females landed at Puerto Cabezas were smaller than the smallest nesting female at Tortuguero in 1988.

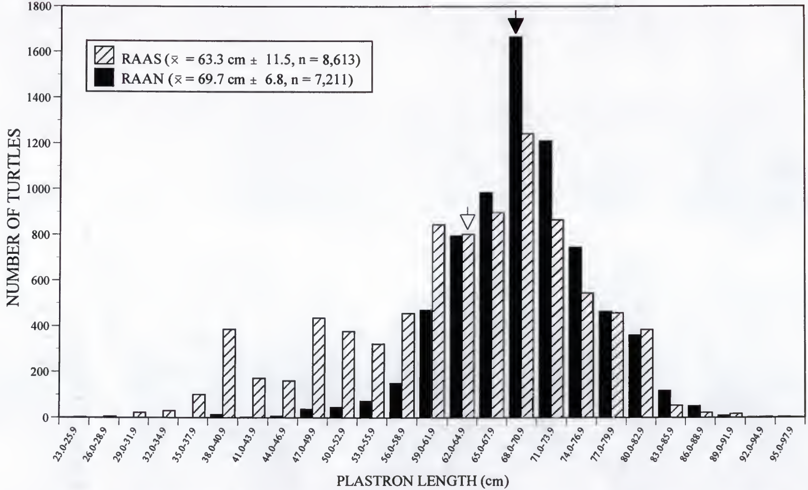


Figure 3.5. Size distribution of plastron length for harvested green turtles, *Chelonia mydas*, by region (Region Autonoma del Atlantico Norte, RAAN; and Sur, RAAS), Nicaragua from February 1994 to December 1996. There was a significant difference between the RAAN and RAAS for plastron length (Wilcoxon Rank Sums, $Z = 34.6$, $P < 0.0001$). Arrows indicate in which size class the mean for each region is located.

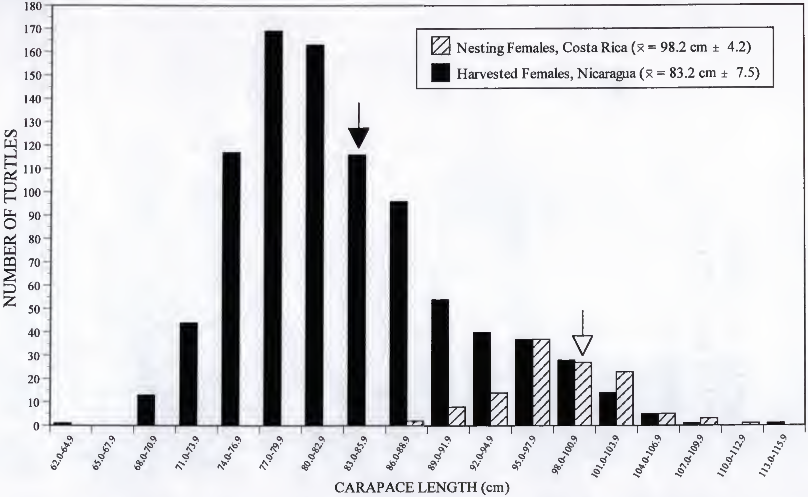


Figure 3.6. Size distribution of minimum straight carapace length (SLN) for harvested female green turtles, *Chelonia mydas*, landed at Puerto Cabezas, Nicaragua from April 1992 to March 1993 and predicted SLN of green turtles nesting at Tortuguero, Costa Rica in 1988 (Tortuguero data courtesy of the Caribbean Conservation Corporation). Predicted carapace lengths are based on the equation Y = -1.358+1.002(SLT), where SLT is the maximum straight carapace length, $r = 0.99$, $P < 0.0001$. Arrows indicate in which size class the mean for each location is located.

Between February 1994 and December 1996, 54.2% of the harvested green turtles measured (n = 15,822 animals) were smaller than the smallest reproductively mature male (SRM; 13.3%) and female (SRF; 40.9%), based on the presence of sperm for males and corpora lutea for females. In the RAAN, of the animals measured, an estimated 4,556 (63.2%) were females, of these 64.9% (2,957) were smaller than the SRF. In contrast, of the estimated 2,653 (36.8%) males, 14.2% (377) were smaller than the SRM. In the RAAS, an estimated 4,530 (52.5%) were females, of these 77.7% (3,518) were smaller than the SRF and for the estimated 4,083 (47.4%) males, 42.3% (1,729) were smaller than the SRM.

Trends in size distribution of harvested animals

Using a regression analysis, mean plastron length (PL) of harvested animals in the RAAN has increased 2.4 cm from February 1994 to April 1997 ($r = 0.10$, $P < 0.0001$; Table 3.3). When measurement data from each site were analyzed separately, however, the change in mean PL of harvested animals for three of the four sites (Puerto Cabezas, Awastara, and Sandy Bay) is ≤ 0.5 cm for the time period ($P \geq 0.12$ for the three sites). At the fourth site (Dakra) there was a 3.5 cm increase in mean PL. However, for animals measured at the commercial center of Puerto Cabezas, where animals captured by turtlers from the three communities, Awastara, Dakra, and Sandy Bay are landed, there was an increase of 0.3 cm in mean PL ($P = 0.27$; Table 3.3).

In the RAAS, mean plastron length of harvested animals decreased 4.6 cm from July 1994 to December 1996 ($r = -0.09$, $P < 0.0001$). At three of the four sites, mean plastron length of harvested animals decreased from 0.3 cm to 8.2 cm during the time

period (Table 3.3). At only one site, Sandy Bay Sirpi, mean plastron length increased 2.7 cm during the time period.

Table 3.3. Simple regression analysis of the change in mean plastron length over time of harvested green turtles, *Chelonia mydas*, from offshore waters of Caribbean Nicaragua. Animals were measured when they were landed at the turtler's community or at the commercial center of Puerto Cabezas. Data were analyzed for each data collection site and by region. RAAN = Región Autónoma del Atlántico Norte) and RAAS = Región Autónoma del Atlántico Sur).

Site or Region	Data Collection Periods (Length of Period)	Change in Mean Size for the Time Period	n	r	P-value
RAAN sites combined	Feb 94 - Apr 97 (1,184 days)	+ 2.4 cm	8,807	0.10	< 0.0001
Awastara	Feb 94 - Jan 95; Dec 95 - Apr 97 (881 days)	- 0.08 cm	1,491	- 0.0	0.85
Dakra	Feb 94 - Jan 95; Dec 95 - Apr 97 (881 days)	+ 3.5 cm	1,341	0.17	< 0.0001
Sandy Bay	Feb 94 - Feb 95; Dec 95 - Apr 97 (909 days)	+ 0.5 cm	2,088	0.03	0.12
Puerto Cabezas	Feb 95 - Apr 97 (819 days)	+ 0.3 cm	3,886	0.017	0.27
RAAS sites combined	Jul 94 - Dec 96 (914 days)	- 4.6 cm	8,371	- 0.09	< 0.0001
Río Grande Bar	Jul 94 - Dec 96 (914 days)	- 0.3 cm	1,426	- 0.01	0.77
Sandy Bay Sirpi	Jul 94 - Dec 96 (914 days)	+ 2.7 cm	2,167	0.10	< 0.0001
Tasbapaune	Jan 95 - Dec 96 (730 days)	- 5.1 cm	4,371	- 0.1	< 0.0001
Set Net	Jul 94 - Dec 96 (914 days)	- 8.2 cm	407	- 0.2	< 0.0001

Hawksbill, Loggerhead, and Leatherback Turtles

Harvest levels

For hawksbills, the minimum harvest in 1994 was 86 animals. In 1995, the minimum harvest was 109 animals and in 1996, it was 53 animals. For loggerheads, the estimated capture in 1994 was 173 animals. In 1995, the minimum number of animals captured was 169, and in 1996, it was 483 animals. From 1994 to 1996, only four leatherbacks, all in 1995, have been reported captured. Three of them were captured by turtlers of Dakra (RAAN) and one by turtlers of Tasbapaune (RAAS). Detailed annual harvest levels for each data collection site by species are located in Appendix F.

Temporal trends in the harvest of hawksbill and capture of loggerhead turtles

For the Caribbean coast of Nicaragua, the mean monthly harvest rate of hawksbills from December 1993 to December 1996 was 6.9 ± 3.9 animals (range = 1 - 18, $n = 37$). Based on a simple regression model, the harvest rate of hawksbill turtles has remained essentially constant during this 3-yr period ($r = -0.23$, $P = 0.17$, $Y = 8.44 - 0.083(ME)$, where ME is the number of months elapsed; Figure 3.7A).

For loggerheads, the mean monthly capture rate from September 1994 to December 1996 was 25.3 ± 17.1 animals (range = 3 - 63, $n = 28$). Based on a simple regression model the capture rate of loggerheads has increased significantly at 17 turtles/yr during this 2.25-yr period ($r = 0.68$, $P < 0.0001$, $Y = 4.80 + 1.42(ME)$, where ME is the number of months elapsed; Figure 3.7B).

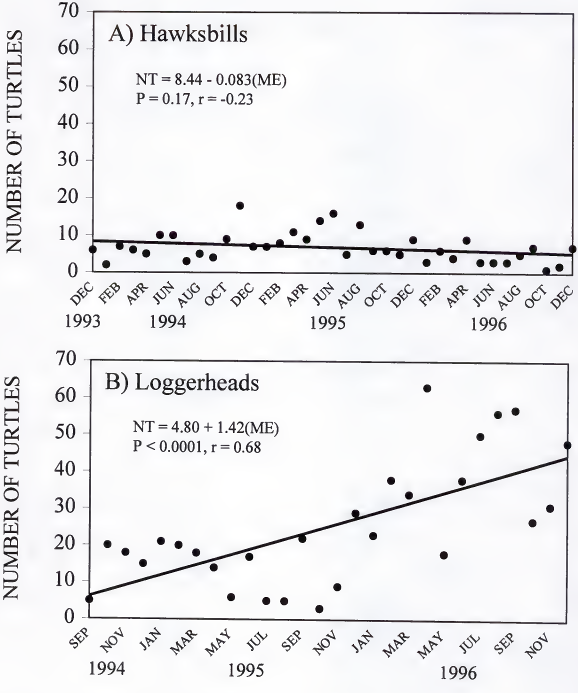


Figure 3.7. Number of A) hawksbill, *Eretmochelys imbricata*, and B) loggerhead, *Caretta caretta*, turtles reported captured from the Caribbean waters of Nicaragua. Hawksbills are reported from December 1993 to December 1996 (37 mo) and loggerheads from September 1994 to December 1996 (28 mo). NT = number of turtles, ME = months elapsed.

Morphometric parameters of harvested hawksbill turtles

Ten body measurements were recorded for six hawksbills. Mean female minimum curved carapace length (CLN) was 77.8 cm ± 7.4 (range = 67.0 - 85.6, n = 5) and the CLN of the male was 73.7 cm. Simple statistics for the ten body measurements are summarized in Appendix G. Because loggerheads are not landed at Puerto Cabezas none were measured.

Demographic parameters of harvested hawksbills

From December 1993 to December 1996, the sex ratio of harvested hawksbills was 1M : 2F (n = 42 animals) for animals with plastrons ≥ 55.9 cm (for animals where sex was determined by external characteristics) or when sex was confirmed during butchering. Sex ratios of harvested animals for the RAAN (n = 24 animals) and RAAS (n = 18 animals) were the same. Because hawksbills are not sexually dimorphic until they approach adult size, a minimum plastron length was used to identify which animals could be accurately sexed based on external characteristics. This cut-off measurement is based on the minimum size (55.9 cm plastron) reported for nesting hawksbills at Tortuguero, Costa Rica (Carr et al. 1966; Bjorndal et al. 1985). Although plastron measurements taken in Nicaragua (curved) and in Costa Rica (straight) were not identical, the difference in these two measurements is probably less than a few centimeters based on plastron length measurements taken from green turtles (C. Campbell unpubl. data).

There was a significant difference in mean plastron length of hawksbill turtles harvested in the RAAN and RAAS (ANOVA, $F_{1,88} = 10.29$, $P = 0.0019$). In the RAAN, between December 1993 and December 1996, mean plastron length was 63.3 cm ± 0.1

(range = 32.0 - 80.7, n = 59; Figure 3.8). In the RAAS, between January 1995 and December 1996, mean plastron length was 55.7 cm ± 11.8 (range = 14.0 - 75.4, n = 31; Figure 3.8).

Overall, regardless of sex, 71.1% of the harvested hawksbills measured (n = 64 animals) were larger than the smallest nesting female (SNF) at Tortuguero, Costa Rica. In the RAAN, 78.0% (46) were larger than the SNF. In contrast, only 58.1% (18) were larger than the SNF in the RAAS.

Demographic parameters of captured loggerheads

Loggerheads are discarded, unconscious or dead, where they are captured, or used for lobster trap or shark bait. Thus, the data collectors had the opportunity to measure only 16 loggerheads. All loggerhead measurements are from the RAAS. Mean plastron length was 62.5 cm ± 10.5 (range = 47.3 - 84.3).

Discussion

Green Turtles

Current and historical harvest levels and trends

For the past six years, the estimated total annual harvest of green turtles from the Nicaragua foraging ground ranged between 6,000 and 11,000 animals. Estimated annual harvests are minimum levels, however, because harvest levels for 1991 to 1993 were based on data collected from 50.0% to 62.5% (4 to 5 sites) fewer sites than during the 1994 to 1996 period. In addition, estimated yearly harvests do not include turtles harvested by three additional turtling communities in the RAAN, or harvests by the Rama

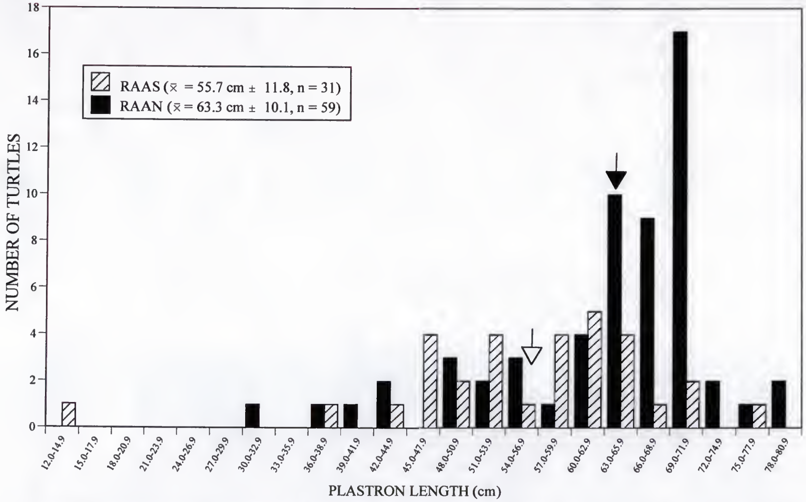


Figure 3.8. Size distribution of plastron length for harvested hawksbills, *Eretmochelys imbricata*, by region (Region Autonoma del Atlantico Norte; and Sur, RAAS) in Nicaragua from December 1993 to December 1996. There was a significant difference between the RAAN and RAAS for plastron length (ANOVA, $F_{1,88} = 10.29$, $P = 0.0019$). Arrows indicate in which size class the mean for each region is located.

Indians in the RAAS. Also, an unknown number of marine turtles are captured incidentally to other fishing activities that occur off Nicaragua's coast, i.e., shrimp trawling, lobster diving, and hook and line fisheries.

In the Región Autónoma del Atlántico Norte (RAAN), the monthly harvest of green turtles for 29 mo between February 1994 and April 1997 has almost doubled since the six-yr period from 1985-1990. The lower harvest rate of green turtles during the 1980s probably resulted from the civil unrest and military conflicts created by the Sandinista/Contra war. According to local informants, many coastal communities were abandoned during the war. In the RAAN, the Sandinista military only allowed fishing trips when they originated from Puerto Cabezas. During times of heightened military activity offshore fishing was either too dangerous or not allowed at all. These factors probably decreased harvest levels of marine turtles as well as terrestrial and other aquatic resources.

Harvest rates of marine turtles during the 1980s (Sandinista/Contra war period) are not available for the Región Autónoma del Atlántico Sur (RAAS). For a post-war period (August 1994 to December 1996), however, monthly harvest rates of green turtles in the RAAS have remained relatively constant. Although monthly harvest rates for the RAAS are available for a relatively short time period, the conclusion that harvest rates have remained stable is also supported by the relatively constant monthly harvest of green turtles by one RAAS community, Sandy Bay Sirpi, for a 6-yr period (1991-1996). Thus, although harvest rates are highly variable (see Figure 3.3), the results suggest that the

overall harvest trend in the RAAS during the 1990s has, thus far, remained relatively unchanged.

Knowledge of historical harvest levels and patterns are important to assessing the possible status of the population and the potential impact of current harvest levels. Unfortunately, no information is available on harvest rates prior to European arrival. As early as 1633, the English established a trading station at Cabo Gracias a Dios, near the Honduras/Nicaragua border. By 1722, Jamaican and possibly Cayman boats were annually visiting the Miskito Cays of Nicaragua to catch and purchase green turtles and hawksbill shell from the Miskitu Indians (Fernández cited in Parsons 1962) and by the early 1800s, Cayman Islanders were regularly turtling off the coast of Nicaragua (Lewis 1940; Parsons 1962). Simmonds (cited in Parsons 1962) reported that by 1878, up to 15,000 turtles annually were landed in Europe, most of them having been caught by the Cayman fleet turtling in Nicaraguan waters. During the first-half of the 20th century approximately 2,000 to 4,000 green turtles were harvested annually from the Nicaraguan coast by Cayman boats (Ingle and Smith 1949; Parsons 1962). From 1958 to 1967, 1,000 to 2,350 green turtles annually were exported from Nicaragua by Cayman boats (Nietschmann 1973). By the mid-1960s, the Nicaraguan government no longer permitted Cayman Islanders to turtle within their waters (Rainey and Pritchard 1972; Nietschmann 1973, 1976). From 1966 to 1976, Nicaragua exported 445,500 kg (equivalent to approximately 10,000 animals) of sea turtle products into the United States alone during 7 of these 10 years (Cato et al. 1978). In the late 1960s and early 1970s, three turtle processing plants began operations in Nicaragua (Nietschmann 1973, 1974).

Unfortunately, prior to the late 1960s, estimates of the harvest were based only on export levels and did not include the harvest of animals for local consumption.

The current annual harvest of approximately 10,000 to 11,000 green turtles is similar to or exceeds annual exploitation levels reported for the first-half of the 1970s (Nietschmann 1973, 1979a, b), when evidence suggesting declines in foraging and nesting populations were attributed by the scientific community to overexploitation (Carr 1969; Nietschmann 1972, 1973, 1974, 1976; Weiss 1976). From 1969 to 1976, during the operation of three turtle processing plants in Nicaragua, there were an estimated 6,000 to 10,000 green turtles harvested annually for exportation and local consumption (Nietschmann 1973, 1979a, b; Bacon 1975). Indications that the foraging ground population was in decline were based on: 1) a decrease in capture rates (in 1971 it took an average two person-days to capture one turtle, and by 1975 it took an average six person-days to capture one turtle, Nietschmann 1973, 1979a; Weiss 1976) , 2) a decrease in the capture of larger turtles (Nietschmann 1972, 1973, 1976), and 3) a severe decline in the 1974 nesting density of females at the Tortuguero, Costa Rica rookery (Carr pers. com. to Nietschmann 1976), the source of most adult turtles on the Nicaragua foraging ground (Carr et al. 1978; Bass et al. in press).

Additional indications that current harvest levels are as high or higher than those in the recent past is demonstrated by comparing past and present harvest levels for two RAAS communities. In Tasbapaune, there has been a two- to three-fold increase in the harvest of green turtles compared to 25 years ago, just prior to the operation of the turtle processing plants. Nietschmann (1972, 1973) reported that 819 turtles were harvested

during a 12-month period beginning in 1968 compared to a range of 1,684 to 2,536 turtles harvested/yr from 1994 to 1996. During the operations of the processing plants, Weiss (1976) reports the harvest level for Sandy Bay Sirpi at 913 turtles during a 12-month period beginning in 1972. For five of the six years from 1991 to 1996, green turtle annual harvest rates ranged from 870 to 1,438, rates which are similar to or exceed the 1972 rate. The early 1970s was a period that, until now, probably had the highest harvest levels to occur on this coast.

The occurrence of several major events in Costa Rica and Nicaragua during the late 1970s and 1980s, probably resulted in a decrease in the harvest rate and aided in some level of recovery of the foraging population. In 1975, the Tortuguero National Park in Costa Rica was established, site of the largest green turtle rookery in the Caribbean, to protect breeding animals in nearshore waters and nesting females and their eggs. This is significant to the Nicaragua foraging population because 1) based on tag recoveries of nesting females from Tortuguero the majority use the Nicaragua foraging grounds (Carr et al. 1978) and 2) based on mitochondrial DNA analysis the majority of subadult and adult animals foraging in Nicaragua are from the Tortuguero rookery (Bass et al. in press). By 1973, the processing plant in Puerto Cabezas was closed, and by 1977, the other two plants in Nicaragua were closed (Nietschmann 1976, 1979b) and Nicaragua became a signatory to the Convention on International Trade in Endangered Species of Wild Fauna and Flora, CITES (Hemley 1994). During the 1980s, the country was involved in a 10-yr long civil war that displaced many people from their Caribbean

lowland communities and greatly reduced the exploitation of terrestrial and aquatic resources (Nietschmann 1995).

These events would certainly have reduced the harvest of marine turtles from the high levels of the late 1960s and early 1970s, providing approximately 15 yrs for the segment of the population targeted in the fishery to increase and could explain why green turtle harvest levels are as high as they are today. During the war years of 1985 to 1988, between 1,600 and 3,000 green turtles were annually landed at Puerto Cabezas, which was the total harvest in the RAAN at the time (Montenegro Jiménez 1992). For 1989 and 1990, when people began to return to their homes and life began to return to some degree of normalcy, 3,200 and 3,380 green turtles, respectively, were landed at Puerto Cabezas, again the total harvest in the RAAN (Montenegro Jiménez 1992). In comparison, between 1994 and 1996, 5,000 to 6,000 green turtles were harvested annually in the RAAN.

Although a law by the central government to protect green turtles has been established, it is ineffective and no cultural taboos or restrictions currently exist within the Miskitu society to protect against overharvesting the resource (V. Renales pers. com.). At present, only inclement weather and holidays limit the Miskitu Indian marine turtle harvest. These limitations do not occur often enough or for sufficient periods of time to diminish overall harvest levels. Almost all monthly harvest totals remain high suggesting that during each month there are a sufficient number of good weather days and few enough holidays to maintain a fairly constant harvest rate. In contrast, Nietschmann (1973) reported that prior to the opening of the turtle processing plants turtlers divided

their time among hunting, turtling, and tending to agricultural plots depending on the season, other household or community demands, and availability of turtles on the foraging ground. For one community, a decrease in monthly turtle harvest levels occurred between April and July and again between September and November (Nietschmann 1973). These declines in harvest levels were attributed to an increase in rainfall exacerbated in the spring months by the temporary emigration of breeding adult turtles from the foraging grounds (Nietschmann 1973). However, when the turtle processing plants opened and the demand for green turtles increased, turtlers extended their turtling activities year around (Nietschmann 1973). In the present study, a decline in the harvest rate did not occur during the months turtles migrate to the nesting beach probably because the majority of animals harvested are subadults and have not yet begun to make seasonal migrations to the nesting beach.

Current local and regional demands for green turtle meat within Nicaragua have grown to equal or exceed export demands for sea turtle products that occurred during the early 1970s. There are also indications that the regional demand for green turtle meat in Nicaragua has not yet been satiated. According to D. Castro (pers. com.), Miskitu Indians from the Río Coco region (located on the border of Honduras) and from the interior areas, who prior to the Sandinista/Contra war did not eat sea turtle meat, have settled in Puerto Cabezas and are now consuming it. In addition, D. Castro (pers. com.) reports that animals are transported by truck from Puerto Cabezas to the Río Coco region where more people are becoming accustomed to eating sea turtle meat, thus, creating a demand where none previously existed.

Demography of the harvest

For three data subsets and two different time periods, the majority (78.2% for Puerto Cabezas, 64.9% for the RAAN, and 77.7% for the RAAS) of harvested females were smaller than the smallest nesting female at Tortuguero indicating that most of the harvested females were immature. For males, the majority (85.8% for the RAAN and 57.6% for the RAAS) of harvested animals were larger than the smallest male observed with sperm in the reproductive tract indicating that most of the harvested males are physiologically mature. The percent of immature animals of both sexes in the harvest is probably higher, however, because the cut-off measurements used to categorize maturity status for males and females were based on the smallest animals of each sex with observed evidence of reproductive maturity. However, not all animals will become sexually mature at the minimum body size (Carr and Goodman 1970; Limpus et al. 1994a, b) nor will all animals breed once they have reached physiological maturity.

When all harvested animals that were measured are considered, regardless of sex, mean plastron length (66.2 cm) is 13.6 cm and 14.6 cm smaller than the mean plastron length of nesting females at the Tortuguero rookery (Carr and Ogren 1960, $\bar{x}$ = 80.8 cm predicted plastron based on maximum carapace measurements; Bjorndal and Carr 1989, $\bar{x}$ = 79.8 cm plastron length). Although plastron measurements reported for Tortuguero animals are straight-line and for Nicaragua animals curved, the difference in these measurements is probably less than 2.0 cm (C. Campbell unpubl. data) and would not account for the approximately 14 cm difference in means reported between the foraging ground and nesting beach.

Previous studies conducted in Nicaragua provide data on the size and sex ratio of harvested animals with which to compare current demographic data of harvested animals. The sex ratios of animals harvested in the RAAN (1M:1.7F) and RAAS (1M:1.1F) for this study are the same as those observed by Mortimer (1981) in March 1975 and June 1976 combined. Carr and Giovannoli (1957) however, report a sex ratio of 1M:2.6F for animals harvested from the Miskito Cay region (RAAN) between February and April 1956. Assuming that the sex ratio of harvested animals is a reflection of the foraging population, these data suggest that the proportion of females in the foraging population in the RAAN decreased between 1956 and 1976. Since 1976, however, the sex ratio in both the north and south regions of the country have remained the same. The decrease in the proportion of females in the harvest since 1956 could reflect the increased susceptibility of females when on the nesting beach and subsequent increased mortality rates. At this time there is no reason to believe that entanglement nets are biased towards the capture of females, however, possible behavioral differences between the sexes, e.g., movement patterns and habitat use, could account for differences in net captures.

The mean weight of animals harvested has, apparently, decreased during the past 20 yrs. The mean live weight (80.6 kg $\pm$ 23.7, n = 1,438) for green turtles landed in Puerto Cabezas from April 1992 to March 1993 was less than the 90.7 kg mean live weight reported for green turtles harvested by Tasbapaune turtlers during a 12-month period beginning in 1968 (Nietschmann 1972, 1973). In addition, turtlers have reported decreasing the mesh size of their nets from a 46-cm bar to approximately 38 - 43-cm bar so that smaller turtles do not escape (V. Renales pers. com.; P. Julias pers. com.). This

suggests turtlers are no longer capturing a sufficient number of larger animals to meet their economic needs and demand for turtle meat.

Trends in the length of captured animals

During this study, for measurements from all turtling communities combined in the RAAN, there has been a significant increase in the length of harvested animals during the past 3.25 yrs. This apparent increase in length of harvested animals during the study period, however, is a result of the data from only one of the four data collection sites included in this analysis. The length of animals from three of the four sites increased ≤ 0.5 cm, while for Dakra, the length of harvested animals increased 3.5 cm. Because turtling communities in the RAAN overlap extensively in their use of capture locations (see Chapter 2), there is no reason to expect the mean length of harvested animals for one community to be different from the other three data collection sites. The apparent increase in the length of animals harvested by Dakra is probably due to errors in measuring or in data recording and, therefore, is probably incorrect. There probably has not been a change in the size of animals harvested in the RAAN during this 3.25-year period.

In the RAAS, overall, there has been a significant decrease in the size of harvested animals during the past 2.5 yrs. The most northern turtling community in the RAAS, however, showed a significant increase in size, while the other three communities showed a decrease in the size of harvested animals. Interestingly, among the three communities that showed a decrease in plastron length, the magnitude of the decrease is larger, from a north to south direction on the turtling grounds. Because turtling communities in the

RAAS rarely overlap in their use of turtle capture locations (see Chapter 2), it is possible that different levels of harvest pressure in the recent past could have impacted areas of the foraging ground differently. We know nothing, however, about movement patterns and habitat use of turtles on this foraging ground.

Hawksbill Turtles

Historical and current harvest levels

Although in recent years the harvest rate of hawksbills from Nicaraguan waters has remained relatively constant (at 6.9 ± 3.9 turtles/mo from December 1993 to December 1996), compared to earlier harvest rates there has been a 10-fold decline. For the late 1960s and early 1970s, Nietschmann (1981) estimated an annual harvest of 1,000 to 1,200 hawksbills from Caribbean Nicaraguan waters. For one community, Tasbapaune, Nietschmann (1972, 1973) reported a harvest of 27 hawksbills from January to June in 1969 and approximately 107 hawksbills for the same six months in 1971. From January to June during this study, Tasbapaune residents harvested 35 hawksbills in 1995, 5 hawksbills in 1996, and 2 hawksbills in 1997. On average, this represents a 479% decrease in the harvest rate of hawksbills by Tasbapaune residents between the late-1960s/early-1970s and the mid-1990s.

The decline in the harvest rate of hawksbills in Nicaragua is probably due to a decrease in the hawksbill population and not a decline in the demand for hawksbill scutes (the source of tortoiseshell). Worldwide, hawksbill populations are in decline due primarily for the demand in tortoiseshell and stuffed animals for the tourist trade (King

1982; Witzell 1983; Milliken and Tokunaga 1987; Bjorndal et al. 1993; Meylan 1997a). Hawksbills together with green turtles were important commodities in trade relationships developed between Nicaragua and the English and later the United States (Nietschmann 1976). At one time, hawksbill shell was a leading export to Europe and the skin was exported to the United States (Nietschmann 1976). Although it has been illegal to export marine turtle products from Nicaragua since 1977, when it became a signatory of CITES (Hemley 1994), there is still a market for hawksbill scutes within Nicaragua.

Tortoiseshell is purchased from fishers by local artisans who make it into various jewelry items. During the 1980s, there were approximately 30 artisans of tortoiseshell in Puerto Cabezas (J. Lackwood pers. com.). Since the departure from Nicaragua of citizens from Soviet-block countries, after the election of the pro-democratic presidency of Chamorro in 1990, the sale of tortoiseshell products in Puerto Cabezas has declined (J. Lackwood pers. com.). By 1992, there were only 17 artisans still working tortoiseshell in Puerto Cabezas (J. Lackwood pers. com.). However, the relatively easy preparation of hawskbill shell for sale or storage, combined with instability in the national economy, and fluctuations in the demand for hawksbill shell encourages people to continue the opportunistic harvest of hawksbills, even though current demand is low. Today, tortoiseshell jewelry is made in both cottage-based industries, as well as, in retail jewelry stores and can be readily found for sale throughout the country. In the Managua international airport, tortoiseshell jewelry is found for sale along with many other tourist items. At the Puerto Cabezas regional airport, local artisans daily display for sale their

tortoiseshell products. In Bluefields, a retail jewelry store sells locally manufactured hawksbill shell products.

Demography of the harvest

The sex ratio of harvested hawksbills is 1M : 2F in the RAAN and RAAS and both juvenile and adult animals are harvested (range = 14.0 - 80.7 cm plastron range). Regardless of sex, the majority (71.1%) were larger than the smallest nesting female (55.9 cm) reported for the Tortuguero, Costa Rica rookery (Carr et al. 1966; Bjorndal et al. 1985). This does not imply, however, that all animals were reproductively mature. Like green turtles, not all hawksbills will become sexually mature at the minimum body size (Limpus 1992a). Although the majority of animals harvested in both the RAAN (78.0%) and RAAS (58.1%) are larger than the smallest nesting female at Tortuguero, the mean plastron length of animals harvested in the RAAN is significantly larger than in the RAAS.

The harvest of hawksbills is probably less selective for a particular size class of animal than the harvest of green turtles because not only are hawksbills captured in nets set for green turtles, but also when encountered opportunistically by lobster divers on the reefs, or when they come ashore to nest (primarily in the RAAS). However, juvenile hawksbills are less well represented in the harvest than might be expected. This might be explained by some combination of the following factors: 1) the large mesh size of the nets selects against their capture; 2) some divers choose not to capture small animals; and 3) there are fewer juveniles in the population because of low recruitment resulting from the overharvest of adults and eggs reported by Nietschmann (1981). Because small

juveniles are known to occur in this area (Nietschmann 1981; D. Castro pers. com.; Lagueux pers. obs.), the low representation of small animals in the harvest is most likely due primarily to fewer juvenile animals in the population.

Loggerhead and Leatherback Turtles

During the past 2.25 yrs, an average of 25.3 loggerheads were captured/mo. The capture rate, however has increased significantly by an average of 17 turtles/yr for this same time period. Since loggerheads are not targeted in the fishery and have little or no economic value, this increase in harvest rate is likely a result of improved reporting by turtlers and data recording by local data collectors. Loggerhead meat has never been in demand for human consumption on the Nicaragua coast; however, over the years, loggerheads, as well as, greens and hawksbills have been harvested for their throat and shoulder skin (Nietschmann 1972, 1981; Bacon 1975) and more recently as bait for the shark and lobster trap fisheries.

Although it is possible to release animals unharmed, most turtlers club them unconscious to facilitate their removal from the nets, prior to release. The use of entanglement nets allows captured animals to be released uninjured, however, the majority, if not all, of the loggerheads captured probably die because the animals are likely to drown when released unconscious. Because the majority of animals are discarded at sea, it is difficult to determine demographics of captured animals. The size range of the few loggerheads measured, however, suggests that both large juveniles and adults use Nicaragua's offshore waters (see Dodd 1988 for a review of loggerhead sizes).

The capture of leatherback turtles by this fishery does not appear to be a problem. Only four animals have been reported captured from 1994 to 1996, although this is probably a minimum level of human-induced mortality. Little is known about the size and sex of leatherbacks that use this habitat. However, turtlers reported that the four captured animals were "large", indicating they were possibly large juveniles or adults. In addition, infrequent nesting of leatherbacks has been reported near Río Grande Bar and hatchlings were observed on the beach in March 1994 (L. Churnside pers. com.)

The importance of the beach and offshore habitats to the survival of loggerhead and leatherback populations in the Caribbean is unknown, and the rookeries of these animals have not been identified. More studies will be needed before we can evaluate the effect of current fishery practices on either of these species.

Conclusions

This study is the first to quantify harvest levels and identify demographic characteristics of harvested turtles along most of the Caribbean coast of Nicaragua. Although only green turtles are targeted in the fishery, three additional species -- hawksbills, loggerheads, and leatherbacks are also impacted by the fishery. Current harvest levels of green turtles are as high or higher than they have probably ever been on this coast. Although harvest levels of hawksbill turtles have declined since the early 1970s, this probably reflects a decline in the population rather than in the demand for tortoiseshell. Prior to this study, no data were available on the capture and occasional use of loggerhead turtles in this fishery. Because loggerheads probably die as a result of

being released unconscious, populations in the region will likely be affected. Thus, it is necessary to determine their natal rookeries to monitor for population trends. It appears that leatherbacks are only rarely captured in the Miskitu Indian marine turtle fishery, and thus, their populations are probably not affected by this fishery.

The majority of green turtles harvested in the fishery are large juvenile females. In contrast the majority of hawksbills harvested are adult females. Insufficient data are available to evaluate the size and sex of captured loggerheads and leatherbacks, although data on the few loggerheads measured and reports from turtlers on leatherbacks indicate large juveniles and adults of both species are captured. If harvest rates are higher than recruitment rates, resulting in declining populations, then clearly the harvest of females could further impede the ability of these populations to recover. Specific recommendations to manage the harvest are provided in Chapter 6.

Marine turtles are highly migratory during several life stages. Therefore, monitoring harvest rates and demographic data of the Nicaragua fishery will aid in evaluating the impact of this fishery on marine turtle populations throughout the greater Caribbean. These data are also necessary to monitor changes in the turtle populations resulting from management strategies or changes in harvest patterns.

CHAPTER 4
REPRODUCTIVE CHARACTERISTICS AND CYCLICITY OF GREEN TURTLES ON A FORAGING GROUND

Introduction

For a species, reproductive success of individuals is critical to the persistence of that species through time. An understanding of an animal's reproductive biology is important to the conservation of the species and can be crucial to properly manage the recovery of threatened or endangered species. In coastal areas of the world, where subsistence or traditional use of sea turtles occurs, an improved understanding of turtle reproductive cycles can aid in developing strategies to reduce detrimental affects of exploitation.

Studies on the reproductive biology of sea turtles have examined a variety of life stages and aspects of their reproductive cycle. The earliest, and to date, the majority of reproductive studies, have focused on the nesting female (Moorhouse 1933; Carr and Giovannoli 1957; Hendrickson 1958; Caldwell 1959; Carr and Ogren 1959, 1960; Carr and Hirth 1962; Carr et al. 1966; Bustard 1972). Studies of nesting females have focused on characterizing intra- and interseasonal reproductive effort by documenting clutch size, number of egg clutches laid per season, relationship between female size and clutch size, and internesting intervals (Pritchard 1969; Schulz 1975; Carr et al. 1978; Hirth 1980).

More recent studies have focused on the endocrinology of nesting females by examining blood levels of gonadotropins and other hormones pre- and post- ovulation and during oviposition (Licht 1980; Licht et al. 1980, 1982, 1985; Wibbels 1988; Rostal et al. 1990; Guillette et al. 1991; Whittier et al. 1997). Blood samples are removed during different stages of the nesting process and hormone concentrations analyzed. A few studies have used laparoscopy as a non-lethal, albeit, invasive method to confirm an animal's sex and to determine reproductive status based on the macroscopic condition of the gonads or through histological examination of gonadal biopsies (Owens 1982; Limpus and Reed 1985a, b; Limpus 1992a; A. Meylan et al. 1992; P. Meylan et al. 1992; Meylan and Meylan 1994; Meylan et al. 1994). Advantages to laparoscopy are that the reproductive system can be evaluated directly, histology can be used to evaluate biopsies of gonadal tissue, animals are not sacrificed, and the same individual can be examined repetitively over time with apparently little or no negative effects (Limpus and Reed 1985a; Meylan *in litt.*). A few studies have reported on morphometrics or seasonal changes in the reproductive system of sea turtles (Aitken et al. 1976; Solomon and Baird 1979; Owens 1980; Licht et al. 1985). However, no study has examined monthly changes throughout the course of a year. The worldwide endangered status of sea turtles prohibits their sacrifice for research. Thus, the study of the morphometrics of the reproductive system is limited to examining dead animals opportunistically, killed incidentally in marine fisheries or by other human activities, or harvested for human use.

The legal harvest of green turtles, *Chelonia mydas*, on the Caribbean coast of Nicaragua by Miskitu Indians for local use provides an excellent opportunity to examine

the reproductive cycle and to determine the reproductive status of harvested marine turtles from a foraging ground. Due to the large number of animals harvested (currently a minimum of 10,000 - 11,000 green turtles annually), there is almost an unlimited supply of samples. Because this fishery occurs on a foraging ground, animals of both sexes, as well as animals in various stages of reproductive immaturity and maturity can be examined.

This study describes the reproductive cycle of male and female green turtles on a foraging ground based on morphometrics and characteristics of the reproductive system. Testes and epididymides of males, and ovaries and oviducts of females were measured. The size of the organs and presence of sperm for males, and the size of the ovaries, characteristics of follicles, and presence of corpora lutea for females were used to describe the reproductive cycle. Reproductive recrudescence was correlated with environmental parameters. The reproductive status of animals harvested in the fishery provides information on the reproductive cycle of green turtles and will be used to aid in evaluating the impact of the current harvest.

Methods

Data Collection

Green turtle reproductive tracts were examined opportunistically when animals landed in Puerto Cabezas, Nicaragua were butchered for sale in local markets (see Chapters 2 and 3 for details on the harvest). In order to include animals from the entire size range of harvested animals, each month I attempted to include a minimum of 10

harvested animals from each of the following four categories: "small" males, "large" males, "small" females, and "large" females. The cut-off between "small" and "large" was based on the size of the smallest nesting females at Tortuguero, Costa Rica. Animals smaller than 89.8 cm minimum straight carapace length (distance between the anterior edge of the nuchal scute and posterior notch) were categorized as "small". I was not always able to meet the minimum monthly sample size for each category because I was limited by the quantity of animals in each category landed in Puerto Cabezas, inaccessibility of several butchers, and the unwillingness of a few butchers to coordinate with me when animals would be butchered.

Animals were measured and weighed the afternoon prior to the night in which they would be butchered. For each turtle, the following measurements were recorded: minimum straight carapace length (SLN) and body mass (WT). See Chapter 3 for a description of measurements. As animals were butchered, I collected testes and epididymides from males, and ovaries and oviducts from females. Measurements of the reproductive tracts were taken as soon as they were removed from the animal. For males, the following measurements were recorded: for each testis, maximum length, maximum width, maximum thickness, wet mass, and volume; and for each epididymis, wet mass and volume.

For females, the following measurements were recorded: for each oviduct, infundibulum, tube, uterus, and total lengths; and for each ovary, wet mass and volume. The infundibulum was measured from the anterior end of the oviduct to the position where the outer walls of the oviduct became parallel. The tube was measured from the

posterior end of the infundibulum to the utero-tubular junction which was identified as a thickening in the oviduct wall. The uterus was measured from the utero-tubular junction to the utero-cloacal junction. In addition, the entire length of the oviduct was measured from the anterior infundibulum to the utero-cloacal junction.

For each ovary, the number of macroscopically visible follicular size classes was recorded and the diameter of 10 representative follicles of the largest follicular size class present were measured. For each ovary, a mean diameter was calculated for the largest size class of follicles present. The presence or absence of corpora lutea were recorded. Length measurements were made with a 150-cm flexible tape measure to the nearest ± 0.1 cm. Testicular width and thickness, and follicular diameter were measured with a dial caliper to the nearest ± 0.05 mm. Mass was measured with spring scales to the nearest ± 0.5 gm and volume was measured by water displacement in a graduated cylinder to the nearest ± 1.0 ml.

For males, minimum size at reproductive maturity was based on the presence of sperm in either the right testis or epididymis, with the exception of two males where the left testis and epididymis were used. A cross-section of testis and epididymis were flushed with a 10% saline solution and the effluent examined under a 400x simple compound microscope for the occurrence of sperm. A testicular sperm density index was developed for animals larger than the smallest animal with sperm. The presence of sperm was ranked based on a qualitative assessment using the following categories: no sperm = 0, 1 sperm = 1, few sperm = 2, some sperm = 3, many sperm = 4, and packed with sperm = 5. Numerical codes for each animal were used to calculate mean sperm density,

rounded to the nearest ± 0.5, for each month. The larger the number, the more dense the number of sperm.

For females, reproductive maturity was based on the number of follicular size classes, color of the largest follicles, and presence of corpora lutea. Color of the largest follicles were categorized as white, light yellow, yellow, or opaque yellow. Follicular color was used as an indicator of vitellogenic status; white indicates pre-vitellogenic, whereas, the three shades of yellow indicate progressively more advanced stages of vitellogenesis. When right and left follicular color differed, the color indicating the more advanced stage of vitellogensis was used. Minimum size at reproductive maturity was determined from examining histograms of the number of follicular size classes, follicular color, and presence of copora lutea with carapace length.

Changes in testicular, epididymal, and ovarian measurements over time were used to determine reproductive seasonality. Only males larger than the smallest animal with sperm and females with a mean follicular diameter ≥ 4.00 mm were used to determine reproductive seasonality. To account for differences in gonadal and epididymal measurements among animals due to different body sizes, a somatic index (SI) was calculated by dividing gonadal or epididymal measurements by live body mass (WT) and multiplying by a constant of 1000.

Reproductive tracts were photographed and fixed in the field in 10% non-buffered formalin and stored in 70% ethyl alcohol at the University of Florida. All Nicaragua and CITES (Convention on International Trade in Endangered Species of Wild Fauna and Flora) permits were obtained.

Statistical Analysis

Right and left gonadal, epididymal, and oviductal measurements of all animals were compared with a paired-difference t-test to determine if there was symmetry among male or female reproductive tissues. When there was no significant difference between right and left measurements a mean of the two measurements was used in subsequent analyses. When significant differences were found between right and left measurements, a Pearson correlation coefficient (PCC) was used to determine if right and left sides varied proportionally. Because sides did vary proportionally, a mean of the two measurements was used for subsequent analyses.

The relationship between month and mean log testicular and epididymal measurements for males, and ovarian wet mass and volume for females was determined using analysis of covariance (ANCOVA), which included carapace length as a covariate (data were log transformed to achieve homoscedasticity, Sokal and Rohlf 1995). The assumption of equal variances among months was tested with Hartley's F_{max} - test (Sokal and Rohlf 1995). For males, the assumption of equal variances was met for all measurements. For females, none of the measurements (diameter of the largest size class of follicles, ovarian wet mass and volume) met the assumption of equal variance. However, Scheffé (1959) has shown that linear relationships, such as analysis of variance (ANOVA) and ANCOVA, are robust without equal variances and thus, the assumption of equal variances for the ANCOVA was ignored. Analysis of variance was used to determine the relationship between month and mean log diameter of follicles in the

largest size class, carapace length was not included as a covariate because it was not significantly correlated with follicular diameter.

Pearson correlation was used to determine if relationships exist between gonadal measurements and the following environmental parameters: mean monthly rainfall, and mean monthly minimum and maximum ambient temperatures. I recorded environmental parameters in Puerto Cabezas during the study period (see Chapter 3).

All statistical analyses were conducted using SAS software (SAS Institute, Inc. 1989). Univariate procedures were used to determine if distributions approximate normality and a t-test was used to test for equality of variances. When assumptions for parametric analyses were not meet and logarithmic transformation of the data still did not meet the assumptions, non-parametric tests were used, except in the case of the ANCOVA analysis. Means ± 1 S.E. are presented.

Results

Males

Between November 1993 and January 1995 (15 mo), 264 male green turtle reproductive tracts were examined (see Table 4.1 for summary statistics). For males, mean minimum straight carapace length (SLN) was 87.0 cm ± 0.4 (range = 68.6 - 100.8 cm, n = 262). Testes varied from oblong to long, thin strips that were either straight or curved. Epididymides tended to be irregular in shape with many lobes. In addition, many testes and epididymides contained various amounts of grey and black pigment.

Table 4.1. Summary statistics for testicular and epididymal measurements for green turtles, *Chelonia mydas*, landed at Puerto Cabezas, Nicaragua from November 1993 to January 1995. Reproductive tracts were examined during each month of the 15-mo study period. Mean minimum straight carapace length for animals included below is 87.0 cm ± 0.4 (range = 68.6 - 100.8 cm, n = 262).

Measurement[a]	Mean ± S.E.	Range	n
TESTES			
Total Length (cm)	22.8 ± 0.30	10.9 - 39.5	257
Maximum Width (mm)	34.30 ±0.70	13.28 - 73.08	259
Maximum Thickness (mm)	6.84 ± 0.24	1.80 - 24.30	260
Wet mass (gm)	33.5 ± 1.9	3.7 - 210.0	258
Volume (ml)	33.7 ± 1.9	3.5 - 200.0	241
EPIDIDYMIDES			
Wet mass (gm)	21.3 ± 0.8	2.4 - 66.0	236
Volume (ml)	20.1 ± 0.8	2.0 - 58.0	236

[a] Mean of right and left measurements.

Right and left measurements for testicular wet mass, volume, maximum length, maximum width, maximum thickness, and epididymal wet mass and volume were significantly different (Paired-Difference t-test; $P \leq 0.025$ for all measurements). Since, right and left measurements were highly correlated (Pearson correlation coefficient, $r > 0.83$, $P < 0.0001$ for all correlations) the mean of right and left testicular and epididymal measurements for each animal was used for all subsequent analyses.

Males with the microscopic presence of sperm ranged from 79.6 cm SLN (harvested in June 1994) to 98.6 cm SLN (harvested in April 1994) (Figure 4.1A). The largest male (98.0 cm SLN) with no evidence of sperm was harvested in October 1994.

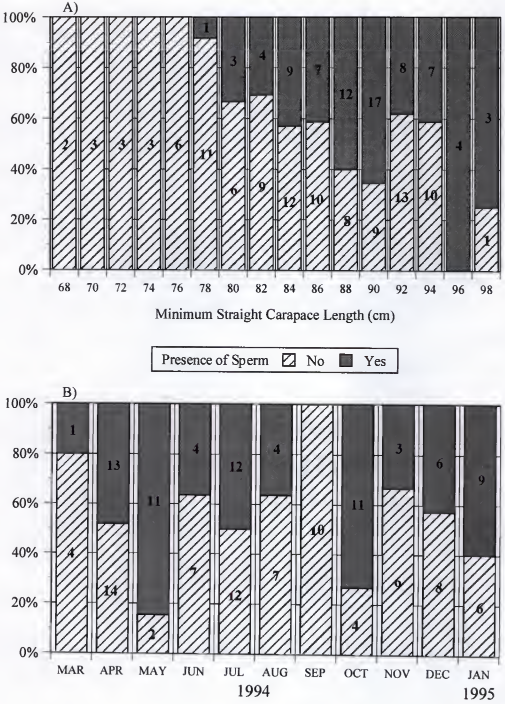


Figure 4.1. Presence of sperm (based on microscopic observation) in the right testis or epididymis of green turtles, *Chelonia mydas,* harvested from northeast Caribbean waters of Nicaragua, by A) minimum straight carapace length (SLN) and B) month of year for males ≥ 80.0 cm SLN. Numbers on bars represent sample sizes. Turtles were landed in Puerto Cabezas, Nicaragua between November 1993 and January 1995.

Males with a SLN $\geq$ 80.0 cm were considered mature, even though sperm were not observed in all of these males (Figure 4.1A). Although one animal smaller than 80.0 cm SLN was observed with sperm, this represented only 8.3% of the animals in that size class, whereas, for all two-cm size classes $\geq$ 80.0 cm SLN more than 30.0% of the males observed had sperm. For males $\geq$ 80.0 cm SLN, sperm were observed in the testis or epididymis of animals harvested during every month of the year except September, though, males harvested in February were not examined for sperm (Figure 4.1B). During months of increased reproductive activity (April, May, and December 1994; and January 1995, see below), sperm were not observed microscopically in 15.4% to 57.1% of the males $\geq$ 80.0 cm SLN (Figure 4.1B).

For mature males (SLN$\geq$ 80.0 cm), an initial wave of testicular recrudescence (based on mean testicular wet mass) occurred in April and peaked in May 1994 (Figure 4.2A and B). A second period of testicular recrudescence began in December 1994 and was continuing when data collection ended in January 1995 (Figure 4.2A and B). In addition, mean monthly testicular sperm density corresponds with changes in testicular size (Figure 4.2). Mean epididymal wet mass did not exhibit seasonality (Figure 4.2A and B). An ANCOVA indicated there was a significant difference among months for mean testicular wet mass, after adjusting for carapace length (ANCOVA, $F_{14, 201} = 2.60$; $P = 0.0018$). Mean testicular wet mass was significantly different between May and September 1994 (Tukey, $P = 0.022$), May and February 1994 (Tukey, $P = 0.0096$), and approaches significance between February 1994 and January 1995 (Tukey, $P = 0.056$). Mean testicular width (ANCOVA, $F_{14, 201} = 1.81$; $P = 0.040$), mean thickness (ANCOVA,

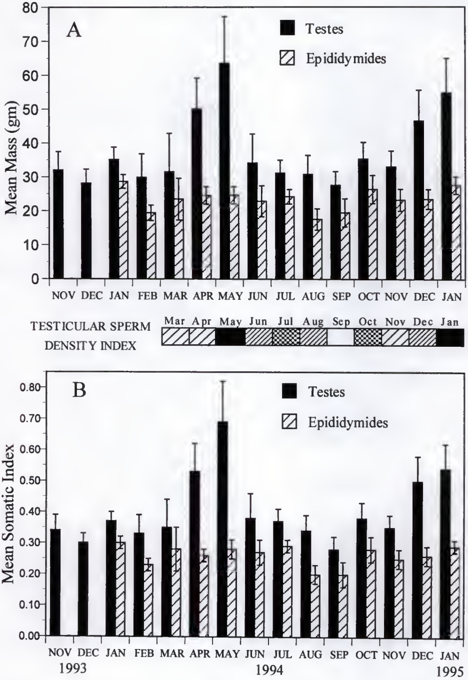


Figure 4.2. Mean (± 1 S.E.) monthly testicular and epididymal A) wet mass and B) somatic indices, for green turtles, *Chelonia mydas*, ≥ 80.0 cm minimum straight carapace length harvested from northeast Caribbean waters of Nicaragua. The testicular sperm density index is based on a qualitative assessment of mean monthly sperm density in the right testis. Increased sperm density is indicated by an increase in shading from light to dark. Turtles were landed in Puerto Cabezas, Nicaragua from November 1993 to January 1995.

$F_{14, 200} = 3.26$; $P < 0.0001$), and mean volume (ANCOVA, $F_{12, 190} = 2.46$; $P = 0.0052$) exhibited a pattern similar to that of mean testicular wet mass (Figure 4.3A and B, testicular volume not shown). Mean testicular length (ANCOVA, $F_{14, 201} = 1.18$; $P = 0.29$) did not show seasonal patterns but remained relatively constant throughout the year (Figure 4.3A and B). Mean epididymal wet mass approached significance between January 1994 and January 1995 (ANCOVA, $F_{12, 187} = 1.76$; $P = 0.058$) and mean volume was significantly different for the same time period (ANCOVA, $F_{12, 188} = 1.95$; $P = 0.031$) (Figure 4.2A and B, epididymal volume not shown). There was a significant difference between the SLN of males without sperm ($\bar{x} = 85.3$ cm ± 1.5, range = 68.6 - 94.6 cm, n = 21) and those with sperm ($\bar{x} = 90.6$ cm ± 0.9, range = 83.4 - 98.6, n = 22) (Wilcoxon Rank Sums, $P = 0.020$) during April and May, months when gonadal size changed the most. However, the size of some males without sperm were larger than those with sperm.

For males with sperm, monthly mean testicular mass (PCC, $r = -0.67$, $P = 0.034$) and volume (PCC, $r = -0.66$, $P = 0.037$) were correlated with decreasing rainfall. Neither monthly mean testicular mass or volume were correlated with monthly mean minimum ambient temperature (PCC, $r = -0.13$, $P = 0.71$ for mass and $r = -0.079$, $P = 0.83$ for volume) or with maximum ambient temperature (PCC, $r = 0.031$, $P = 0.93$ for mass and $r = 0.050$, $P = 0.89$ for volume).

Five males with apparently functional reproductive tracts were observed with incomplete oviducts. Mean SLN for these males was 91.8 cm ± 2.5 (range = 86.7 - 98.1 cm, n = 4). Their oviducts varied from a short segment to several non-continuous

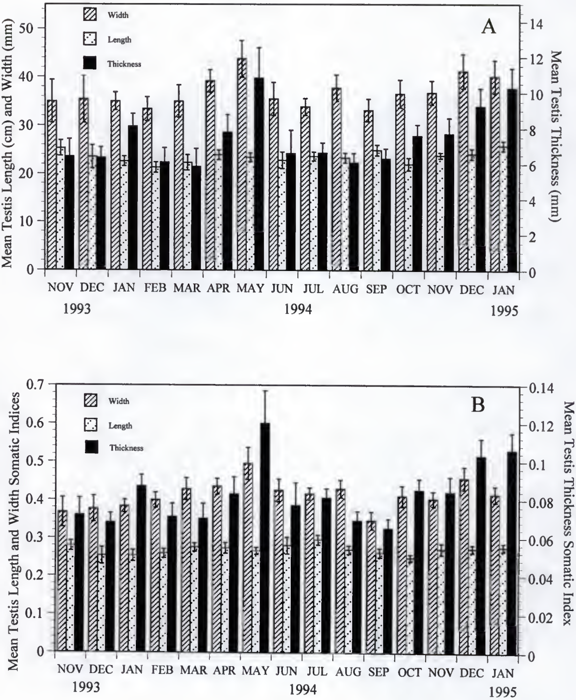


Figure 4.3. Mean (± 1 S.E.) monthly testicular measurements of green turtles, *Chelonia mydas*, ≥ 80.0 cm minimum straight carapace length by A) maximum length, width, and thickness and B) somatic indices. Turtles were harvested from northeast Caribbean waters of Nicaragua and landed in Puerto Cabezas, Nicaragua from November 1993 to January 1995.

segments with each ending in a blind sac. Sperm was observed in two of the three animals examined; one of these contained sperm in only the testis, and the other contained sperm in both testis and epididymis. Testicular and epididymal measurements for these males were within the range of measurements recorded for males without female reproductive tissue. Testicular measurements were as follows: $\bar{x}$ length = 25.1 cm ± 2.1 (range = 21.5 - 28.6 cm, n = 3), $\bar{x}$ width = 40.38 mm ± 4.76 (range = 31.20 - 47.13 mm, n = 3), $\bar{x}$ thickness = 11.90 mm ± 3.07 (range = 6.08 - 16.50 mm, n = 3), $\bar{x}$ wet mass = 57.2 gm ± 19.1 (range = 20.5 - 84.3 gm, n = 3), and $\bar{x}$ volume = 54.2 ml ± 17.8 (range = 20.0 - 80.0, n = 3). Epididymal measurements were as follows: $\bar{x}$ wet mass = 36.4 gm ± 10.0 (range = 23.3 - 56.0 gm, n = 3) and $\bar{x}$ volume = 24.8 ml ± 3.3 (range = 21.5 - 28.0, n = 2).

Females

Between November 1993 and January 1995 (15 mo), 265 female green turtle reproductive tracts were examined (see Table 4.2 for summary statistics). For females, mean minimum straight carapace length (SLN) was 88.3 cm ± 0.5 (range = 70.5 - 107.5, n = 261). Ovaries varied from a firm to watery and flaccid texture, and the color varied from pink to transparent. Oviducts varied in color from white to pink. In addition, many oviducts had various amounts of grey and black pigment on the outer surface primarily along the tube and posterior uterus portions. No pigmentation was observed on the ovaries. There was no difference between right and left measurements for tube length, total oviductal length, and mean diameter of the largest size class of follicles (Paired-Difference t-test; P > 0.24 for each) and bordered on significance for uterus length

Table 4.2. Summary statistics for ovarian and oviductal measurements for green turtles, *Chelonia mydas*, landed at Puerto Cabezas, Nicaragua from November 1993 to January 1995. Reproductive tracts were examined during each month of the 15-mo study period. Mean minimum straight carapace length for animals included below is 88.3 cm ± 0.5 (range = 70.5 - 107.5 cm, n = 261).

Measurement[a]	Mean ± S.E.	Range	n
OVARIES			
Diameter of the largest size class of follicles (mm)[b]	3.85 ± 0.12	0.98 - 20.20	261
Wet mass (gm)	64.4 ± 4.3	5.3 - 435.0	256
Volume (ml)	63.7 ± 4.5	5.0 - 407.5	230
OVIDUCTAL LENGTHS			
Infundibulum (cm)	8.1 ± 0.5	0.5 - 37.0	251
Tube (cm)	156.5 ± 7.9	25.0 - 498.0	220
Uterus (cm)	87.1 ± 4.2	7.8 - 340.5	226
Total (cm)	234.1 ± 12.1	43.0 - 903.5	247

[a] Mean of right and left measurements.
[b] Based on the mean of ten representative follicles.

(Paired-Difference t-test; $P = 0.068$). Right and left measurements for ovarian wet mass and volume, infundibulum length, and mean number of follicular size classes were significantly different (Paired-Difference t-test; $P < 0.022$ for each). Since right and left measurements were highly correlated (PCC, $r > 0.95$, $P < 0.0001$ for each), I used the mean of right and left ovarian and oviductal measurements for each animal for all subsequent analyses.

Mean number of macroscopic size classes of follicles ranged from zero to seven ($\bar{x} = 3.1 \pm 0.05$, $n = 261$). Ovaries of the majority (59.9%) of females had a mean of three

size classes of follicles, and all but three females had a mean of at least two follicular size classes. Ovaries of smaller females had fewer size classes of follicles than larger females, although none of the largest animals had the greatest number of follicular size classes (Figure 4.4). Ovaries of females < 85.0 cm SLN contained from no macroscopically visible follicles to a mean of four follicular size classes. Females ≥ 91.0 cm and < 101.0 cm SLN contained up to seven size classes of follicles (Figure 4.4).

Larger females exhibited more advanced stages of vitellogenesis (Figure 4.5A). Females < 75.0 cm SLN did not exhibit vitellogenic activity and none of the females < 83.0 cm SLN exhibited the most advanced stages of vitellogenesis based on the color of the largest size class of follicles. However, females that were ≥ 83.0 cm SLN were in varying stages of vitellogenesis, as well as, non-vitellogenic (Figure 4.5A).

Only two of the 265 females examined contained shelled eggs in their oviducts. One female, landed at Puerto Cabezas on 17 November 1993, contained eight shelled eggs at the distal end of the left oviduct. Egg diameters ranged from 41.60 to 45.10 mm (43.14 mm ± 1.15, n = 8). The other female, landed at Puerto Cabezas on 26 July 1994, contained one shelled egg in the lower right oviduct. The diameter of this egg was 43.25 mm. The egg shell surface was rough and crusty.

A greater percentage of the larger females have macroscopic copora lutea, evidence that they have ovulated. No corpora lutea were observed macroscopically on ovaries of females < 87.0 cm SLN and as large as 100.9 cm SLN indicating ovulation had

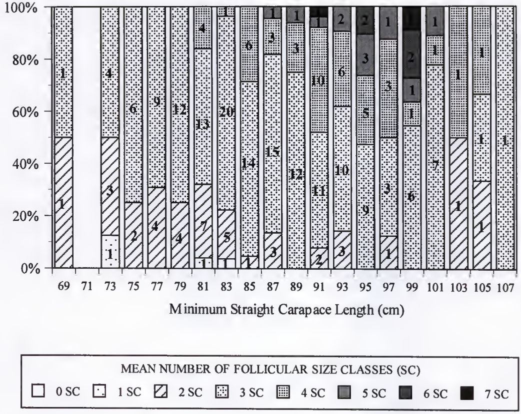


Figure 4.4. Mean number of follicular size classes of right and left ovaries by minimum straight carapace length of green turtles, *Chelonia mydas*. Animals were harvested from northeast Caribbean waters of Nicaragua and landed in Puerto Cabezas, Nicaragua from November 1993 to January 1995. Numbers on bars represent sample sizes.

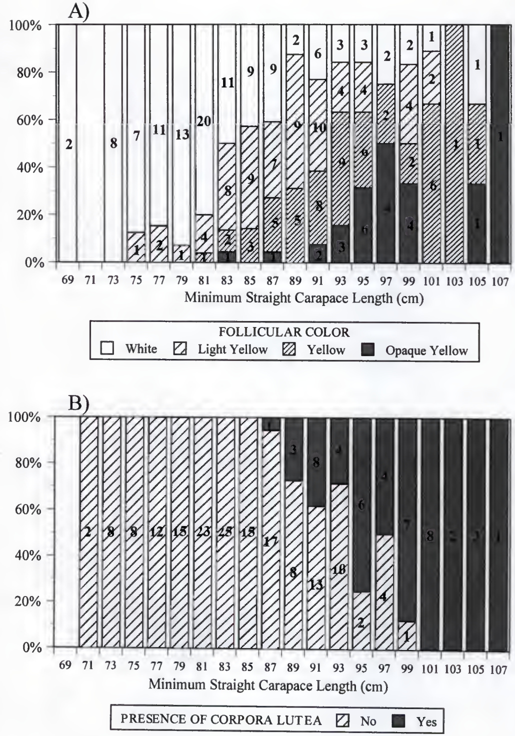


Figure 4.5. Follicular color (A) and presence of corpora lutea (B) for the right and left ovaries of green turtles, *Chelonia mydas*, by carapace length. Turtles were harvested from northeast Caribbean waters of Nicaragua and landed in Puerto Cabezas, Nicaragua from November 1993 to January 1995. Numbers on bars represent sample sizes.

not occurred, at least not recently (Figure 4.5B). All ovaries of animals ≥ 101.0 cm and < 109.0 cm SLN had corpora lutea.

Based on an examination of the number of follicular size classes, follicular color, and presence of corpora lutea by carapace length, nearly all females < 89.0 cm SLN were reproductively immature. Therefore, reproductive seasonality based on the mean number of follicular size classes and follicular color was examined using only females ≥ 89.0 cm SLN. Mean number of follicular size classes varied seasonally, although in all months females had at least two follicular size classes, the majority (52.2%) had three (Figure 4.6A). In April 1994, 36.4% and in January 1995, 60.0% of the females had ovaries with five to seven follicular size classes.

Based on follicular color, mature females are in varying degrees of vitellogenesis each month of the year (Figure 4.6B). Although a highly distinct pattern does not appear, females with follicles in more advanced stages of vitellogenesis occurred between July and October 1994. Females with the least reproductive activity occurred in the months of February, May, and June 1994.

To examine reproductive seasonality based on mean diameter of the largest size class of follicles, and ovarian wet mass and volume, only females with a mean follicular diameter ≥ 4.00 mm for the largest size class (Size Class I) were examined (based on Limpus and Reed 1985a). Ovarian recrudescence (based on mean diameter of Size Class I follicles, and mean ovarian wet mass and volume) occurred in March and peaked in April 1994. A second period of ovarian recrudescence began in December 1994 and was

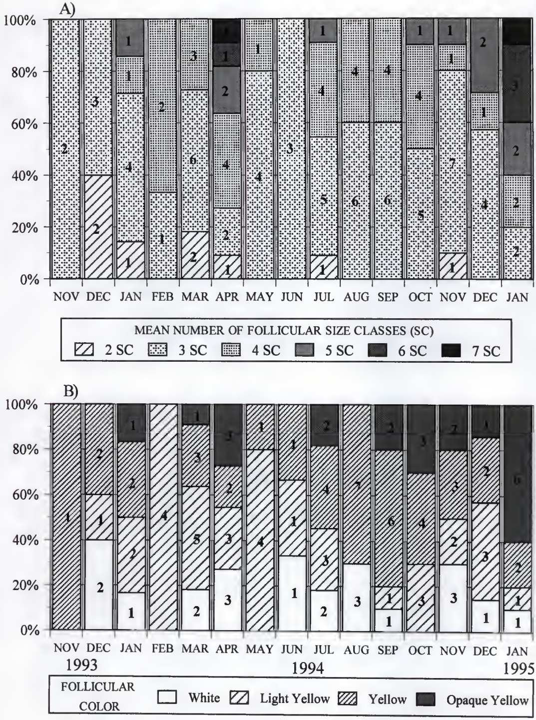


Figure 4.6. Mean number of follicular size classes (A) and follicular color (B) by month for green turtles, *Chelonia mydas*, $\geq$ 89 cm minimum straight carapace length. Turtles were harvested from northeast Caribbean waters of Nicaragua and landed in Puerto Cabezas, Nicaragua from November 1993 to January 1995. Numbers on bars represent sample sizes.

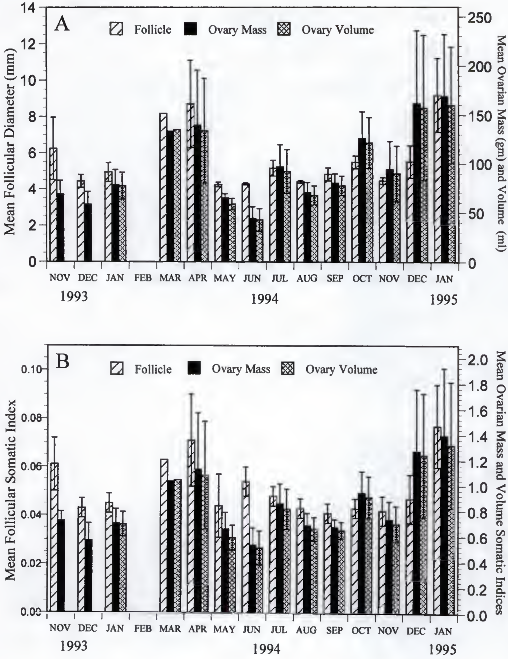


Figure 4.7. A) Mean (± 1 S.E.) monthly diameter of the largest follicular size class, mean ovarian wet mass and volume and B) somatic indices, for green turtles, *Chelonia mydas*, with the largest size class of follicles ≥ 4.0 mm. Turtles were harvested from northeast Caribbean waters of Nicaragua and landed in Puerto Cabezas, Nicaragua from November 1993 to January 1995. Standard error bars are shown when sample size is greater than one.

continuing when data collection ended in January 1995 (Figure 4.7A and B). An ANOVA indicated there was a significant difference among months for the mean log diameter of Size Class I follicles (ANOVA, $F_{13, 55} = 2.21$; $P = 0.021$). Mean log diameter of Size Class I follicles was significantly different between August 1994 and January 1995 (Tukey, $P = 0.047$) and approached significance between November 1994 and January 1995 (Tukey, $P = 0.052$). Although mean log ovarian wet mass and volume exhibited a similar reproductive cycle as follicular diameter, an ANCOVA indicated there was no significant difference among months for either ovarian wet mass (ANCOVA, $F_{13, 52} = 0.78$; $P = 0.68$) or volume (ANCOVA, $F_{11, 48} = 0.81$; $P = 0.63$).

Mean diameter of the largest size class of follicles (for females with follicular diameter ≥ 4 mm) was correlated with decreasing monthly rainfall (PCC, $r = -0.67$, $P = 0.035$). Mean ovarian wet mass and volume (PCC, $r = -0.70$, $P < 0.024$, for each) were correlated with decreasing monthly mean minimum ambient temperature. Mean diameter of Size Class I follicles was not correlated with either monthly minimum (PCC, $r = -0.35$, $P = 0.32$) or maximum (PCC, $r = -0.18$, $P = 0.62$) ambient temperatures. Monthly mean wet mass and volume were not correlated (PCC, $r < -0.56$, $P > 0.09$, for each) with either monthly rainfall or maximum ambient temperature (PCC, $r < -0.51$, $P > 0.14$, for each).

Discussion

Based on tag recoveries (Carr et al. 1978) and mitochondrial DNA (mtDNA) analysis (Bass et al. in press) the principal rookery, for adult-size green turtles found on the Nicaragua foraging ground is Tortuguero, Costa Rica. The green turtle nesting season

at Tortuguero is from June to October with peak nesting from July through September (Carr et al. 1978). Mating occurs during nesting migrations (P. Meylan et al. 1992) and in the vicinity of the nesting beach (Carr 1954, 1956; Carr and Ogren 1960; Carr et al. 1978; Mortimer 1981; Hirth 1997). Although specific migratory movements have not been described, reproductively active males and females occur in waters off the nesting beach (Carr 1956; Carr and Ogren 1960; Carr et al. 1978; Ross and Lagueux 1993; Hirth 1997); thus, each season, both male and female breeding animals leave the foraging ground.

Not all adult-size males on the Nicaragua foraging ground are annual breeders, although they can breed annually (Balazs 1983; Limpus 1993). Males as large as 98.0 cm (minimum straight carapace length, SLN) had no microscopic evidence of sperm. Not all males ≥ 80.0 cm SLN had sperm during months of reproductive activity nor did any of the 10 animals (SLN ≥ 80.0 cm) examined in September, suggesting that none of these males bred during the current reproductive season. Recapture intervals of 1-5 yr of tagged males in the Hawaiian Islands (Balazs 1983) and in Australia (Limpus 1993) also suggest that males are not annual breeders. However, not all males attain sexual maturity at a minimum size (Limpus 1990, Limpus et al. 1994a, b). Determining breeding intervals based on recaptures of tagged animals should be viewed with caution because tag loss can be high (Limpus 1992b; Bjorndal et al. 1996) and recapture rates are not 100%. Identifying the proportion of annual breeders in a population is necessary to avoid overestimating the size of the breeding population.

Seasonal reproductive activity was not observed in all testicular, epididymal, ovarian, and oviductal measurements. For males, the greatest seasonal change occurred in mean testicular mass, volume, and thickness. Mean testicular length, and epididymal mass and volume were the poorest indicators of seasonality. For females, the greatest seasonal change occurred in the mean diameter of the largest size class of follicles (for females with follicles $\geq$ 4.00 mm). Mean ovarian mass and volume also exhibited seasonal changes, but not to the same extent as follicular diameter.

Right and left measurements of tube length, total oviductal length, and mean diameter of the largest size class of follicles did not differ significantly. Thus, either right or left measurements would be sufficient for future studies. All other measurements differed significantly, ovarian wet mass and volume; infundibulum and uterus lengths; number of follicular size classes; testicular wet mass and volume; maximum testicular length, width, and thickness; and epididymal wet mass and volume. Although they differed significantly, they did so proportionally, within the size range of the animals examined. Therefore, it is necessary to either measure both sides of the animal and report the mean, or, if only one side is measured, the side measured should be consistently reported. Otherwise, comparisons of reproductive tract measurements should not be made among studies.

The synchronization in the peak and nadir of reproductive activity throughout the year for males and females is very similar. Green turtles in Nicaragua have a prenuptial cycle, as previously reported for captive males (Licht et al. 1985), migrating males (Engstrom 1994) and loggerheads, *Caretta caretta*, (Wibbels 1988; Wibbels et al. 1990).

That is, sperm production and vitellogenesis are completed just prior to mating (Licht 1984). The annual gonadal cycle agrees with the plasma testosterone cycle of a captive breeding colony (Licht et al. 1985). This contrasts with the postnuptial cycle reported for many temperate species of freshwater turtles (Moll 1979; Licht 1982). Measures of ovarian activity (i.e., mean follicular diameter, ovarian mass and volume) peaked in April for females, whereas, testicular activity (i.e., mean mass, volume, width, and thickness) peaked in May for males. Both ovarian and testicular measurements were increasing in December 1994 and January 1995, when the study ended, at levels equal to the April and May 1994 peaks. Due to a sharp decrease in gonadal measurements, it appears that reproductively active females left the northeast foraging ground of Nicaragua for the nesting beach at Tortuguero in May, whereas, males left in June. It is unknown, however, if males and females arrive separately at the nesting beach, rejoin each other along the migratory route, or migrate separately but synchronize their arrival at the nesting beach (Hirth 1997). Because only animals from the northern extent of the Nicaragua foraging area were examined, it is possible that animals foraging on the southern Nicaragua grounds (i.e., located offshore between the communities of Sandy Bay Sirpi and Set Net, see Figure 2.1) show a different migratory schedule.

During this study, gonadal recrudescence occurred at two different time periods for both males and females. Gonadal size peaked in April 1994 for females and in May 1994 for males, and was increasing for both sexes in December 1994 and January 1995. It is notable that the increase in gonadal activity that occurred in December 1994 and January 1995 for both sexes was not observed during the beginning of the study in

December 1993 and January 1994. The difference observed in the onset of gonadal activity during the study can be hypothesized to be due to a shift in reproductive recrudescence as a response to a change in environmental cues (Duvall et al. 1982; Crews and Gans 1992; Whittier and Tokarz 1992) or due to sampling animals from rookeries with different nesting seasons.

Yearly shifts in the onset of reproductive activity can be caused, in part, by variability in the onset of environmental parameters from year to year, e.g., precipitation and temperature. There is a significant correlation between the El Niño Southern Oscillation (ENSO) and the number of nesting females at Heron and Raine Islands, Australia approximately two years later (Limpus and Nicholls 1988). The ENSO is a combination of pressure, temperature, and rainfall fluctuations. In Nicaragua, testicular mass and volume, and diameter of the largest follicular size class were correlated with decreasing rainfall, and ovarian mass and volume were correlated with decreasing minimum ambient temperature. Because exogenous factors affect seasonal movements on the grassbeds, they can also play a role in stimulating or delaying reproductive activity. During the rainy season turtles forage on grassbeds located further from shore to avoid the discharge of silt laden freshwater from numerous river mouths that can extend 16 to 18 km from the coast (Nietschmann 1973; Mortimer 1981). Therefore, changes in the onset of environmental cues, e.g., rainy season, can cause a shift in reproductive recrudescence possibly accounting for the difference observed between the years.

Another explanation for observing reproductive activity during different times in the study period could be due to examining animals from rookeries with different nesting

seasons. Based on tag recoveries and mtDNA analysis of adult-size males and females, the Nicaragua foraging ground population comprises animals from at least two different rookeries, Tortuguero Costa Rica (Carr and Ogren 1960; Carr et al. 1978; Bass et al. in press), and Aves Island, Venezuela (Solé 1994; Bass et al. in press). The green turtle nesting seasons for these rookeries are: June to October for Tortuguero (Carr et al. 1978) and February to November for Aves Island (Licht et al. 1980; Pritchard 1984; Solé and Medina 1989). The reproductive activity observed for some of the animals harvested in December 1994 and January 1995 could have been from the Aves Island rookery whose nesting season begins four months prior to the nesting season at Tortuguero. The absence of reproductively active animals in the December 1993 and January 1994 samples can be explained by the low probability of sampling animals from rookeries other than Tortuguero, such as Aves Island, because 1) male and female green turtles are not annual breeders (Carr et al. 1978; this study) and thus, there is a small proportion of mature animals reproductively active each season on the foraging ground, and 2) the northern extent of the foraging ground is comprised of a high proportion (90%) of adult Tortuguero animals (Bass et al. in press), further reducing the probability of sampling reproductively active Aves Island turtles.

Few studies have reported on the morphometrics of organs and structures associated with sea turtle reproductive systems (Aitken et al. 1976; Solomon and Baird 1979; Owens 1980; Licht et al. 1985), and fewer have described seasonal morphometric changes with which comparisons can be made (Licht et al. 1985). In this study, mean diameter of the largest follicular size class (20.2 mm) did not approach the size of

follicles measured just prior to ovulation (30 - 35 mm; Aitken et al. 1976; Solomon and Baird 1979). This suggests that females depart the northern extent of the Nicaragua foraging ground toward the nesting beach prior to ovulation. This hypothesis is further supported by the absence of eggs in the oviducts. Only two females were observed with one and eight shelled oviductal eggs. These eggs were probably retained from a previous nesting event as opposed to being shelled in preparation for an upcoming nesting event. Egg diameters (41.60 - 45.10 mm) were within the range reported for eggs laid at the Tortuguero, Costa Rica rookery (range = 39.1 - 48.4 mm, Bjorndal and Carr 1989).

None of the testes observed during this study attained a mass equal to two of the three (individual animals) testes observed from green turtles harvested nearly twenty years ago from the same area in Nicaragua (Owens 1980). The largest testicular mass I recorded was 210.0 gm, which occurred in April 1994, compared to 237.1 gm and 365.9 gm, also for April 1976, reported by Owens (1980, *in litt.*). Using a gonadal somatic index (GSI) to account for the effect of carapace length on testicular mass, the males observed by Owens (1980) had GSIs of 3.42 and 2.35 compared to 2.18 and 1.88 for the two males with the largest testicular mass and GSIs observed in this study. First-time nesting females lay fewer clutches (Limpus 1996) and fewer eggs per season (Carr and Ogren 1960; Carr et al. 1978; Fowler 1979; Frazer 1984; Limpus 1996) than more experienced nesters. If males follow the same pattern as females, then the lower GSIs (compared with Owens 1980) observed in this study could indicate that the more experienced breeders are less abundant than 20 years ago (Owens *in litt.*).

Conclusions

The Nicaragua green turtle fishery includes large juveniles, novice and mature breeders of both sexes. The life stages targeted by the fishery are the most valuable animals to a population in terms of maintaining population stability. For unharvested populations, survival rates of large juveniles and adults of long-lived, slow-maturing, iteroparous species are necessarily high to maintain stable populations (Congdon and Dunham 1994). Green turtles can take from 20 to 60 years to reach sexual maturity (Limpus and Walter 1980; Mendonça 1981; Burnett-Herkes et al. 1984; Frazer and Ehrhart 1985; Zug and Balazs 1985; Frazer and Ladner 1986). The majority of animals captured in the fishery are harvested after they have almost survived the 20 or more years needed to attain reproductive maturity, thus eliminating any reproductive and genetic contributions they could have made to future populations.

Many nesting studies (Carr et al. 1978; Balazs 1980, 1983; Limpus and Reed 1985a; Mortimer and Carr 1987; Hirth 1997 for a review) have confirmed that females take multiple years between nesting events and it appears that at least some males can also take more than one year between breeding events (Balazs 1980, 1983; Limpus 1993; this study). In addition, more experienced breeding males could be less frequent than in the past, suggesting that this segment has been harvested out of the population. The fishery should be managed to allow as many individuals as possible to reach full reproductive maturity.

If a green turtle harvest is to continue, then it must be understood that the fishery removes the animals most valuable to the population as indicated by the life stages targeted in the fishery (this study) and their apparent importance to population growth (Crouse et al. 1987; Congdon et al. 1993, 1994; Congdon and Dunham 1994; Crowder et al. 1994; Heppell et al. 1995; Heppell et al. 1996a, b). More research is needed to thoroughly evaluate the impact of the fishery on the turtle population, however, measures should be adopted now to mitigate possible impacts to the turtle population. One measure would be to lower the annual harvest levels of green turtles. Another measure would be to regulate the size and sex of animals harvested. Although all animals harvested are valuable to population growth it might be less detrimental to the population to harvest a larger proportion of males and specifically, harvest the larger males. Because sea turtles are polygynous, one male can fertilize several females (Limpus 1993). By harvesting larger males this would allow them the opportunity to mate and pass on their genetic lineage prior to being harvested, however selective harvesting as suggested here should be approached cautiously. Harvest levels will need to be regulated and monitored because we lack knowledge about the role of accessory males in accompanying or attending mating pairs (Carr 1956; Carr and Giovannoli 1957; Hendrickson 1958; Booth and Peters 1972; Balazs 1980; Limpus 1993). It is likely that a combination of strategies will be needed to regulate the Nicaragua harvest and maintain a viable foraging population of green turtles for future generations of coastal residents.

CHAPTER 5
ASSESSMENT OF HARVEST LEVELS AND THEIR IMPACT ON MARINE TURTLE POPULATIONS

Introduction

The Need to Evaluate the Nicaragua Marine Turtle Fishery

Worldwide, marine turtles are endangered due to hundreds of years of harvesting animals and their eggs (see review of worldwide exploitation in Chapter 1), primarily for meat and tortoiseshell, but also for oil and calipee, and most recently due to habitat destruction or alteration, and incidental capture (Parsons 1962, 1972; Bjorndal 1982; National Research Council 1990; Lutcavage et al. 1997). Throughout the greater Caribbean, green (*Chelonia mydas*) and hawksbill (*Eretmochelys imbricata*) turtle populations have declined from the abundant levels described by early Europeans on their arrival to the New World (Carr 1954; Parsons 1962, 1972; Bjorndal et al. 1993; Meylan 1997a). Today, Nicaragua has the largest remaining green turtle foraging population in the Atlantic Ocean (Carr et al. 1978).

Juvenile and adult green turtles tagged in at least eight countries (Solé 1994; Bjorndal and Bolten 1996; Bagley *in litt.*; Bresette *in litt.*; Meylan *in litt.*; Ehrhart pers. com.; Moncada pers. com.; Lagueux pers. obs.), and hawksbills tagged in at least three countries (Carr et al. 1966; Carr and Stancyk 1975; Bjorndal et al. 1985; Hillis 1994;

Garduño *in litt.*; see Meylan 1997b for review) throughout the greater Caribbean have been harvested in Nicaragua waters. Two loggerheads (*Caretta caretta*) one tagged in Panama (Meylan *in litt.*) and one in the Azores, Portugal (Bjorndal *in litt.*) were also recovered in Nicaragua. Because sea turtles move through several developmental habitats (Musick and Limpus 1997) and adults migrate between nesting and foraging grounds (Meylan 1982), the Nicaragua marine turtle fishery can impact turtle populations shared by several nations, across great distances. In addition, conservation efforts enacted by other nations in the Caribbean to protect sea turtles could be diminished or nullified by an overharvest of turtles in Nicaragua.

The central government has established seasonal regulations to control the harvest of green turtles off Nicaragua's Caribbean coast, however, they are unclear, unenforced and ineffective (Nietschmann 1973; Peralta Williams 1991; D. Castro pers. com.; C. Lagueux pers. obs.). The closed season has at different times varied in duration and included a total ban against the harvest of turtles, a ban against the commercialization of marine turtles, and a ban against the harvest of females (Nietschmann 1972, 1973; Weiss 1976; Montenegro Jiménez 1992). Despite these regulations the turtle harvest continues unabated throughout the year. The turtle fishery is neither managed by the central or regional governments nor by Miskitu Indian taboos or restrictions against the harvest or consumption of marine turtles (D. Castro pers. com.; M. Grantt pers. com.; V. Renales pers. com.). It is imperative to evaluate this fishery and its possible impact on marine turtle populations in the region because: 1) sea turtles are an important economic and cultural resource to the Miskitu Indians; 2) the Nicaragua green turtle foraging population

is the largest in the western Atlantic (Carr et al. 1978); 3) sea turtles are a shared international resource; 4) evidence from elsewhere suggests that sea turtles can be readily fished to extinction; and 5) there are no provisions currently in Nicaragua to protect against overharvest.

Approaches to Evaluating Biological Sustainability

There are several approaches to evaluating the biological sustainability (i.e., the ability of the population to maintain itself over time) of harvesting a resource such as marine turtles. Monitoring population size for changes caused by harvesting is one approach that requires a comparison of population estimates at different time periods. Population size or density is usually determined by sampling some portion of the population because a complete population census is rarely possible. Population estimates can be based on direct or indirect sampling (Caughley 1977; Davis and Winstead 1980; Caughley and Sinclair 1994). Common direct sampling methods include counting animals observed on transects of known area and through mark/recapture methods. Indirect sampling methods use evidence of animal presence, e.g., counting tracks, scats, or calls, per a unit of area or time. Furthermore, the population's rate of increase (r) can be determined from these population estimates. The resulting r indicates if the population is increasing, decreasing, or stable and thus, can provide an indication of overharvest (Caughley and Gunn 1996). An alternative method to estimating r is through modeling population behavior based on demographic parameters of the population

(Caughley and Gunn 1996). However, the data needed (e.g., survival rates, fecundity, and population counts) to determine *r* by either method can be difficult to obtain.

Another approach to assessing biological sustainability is to compare a population's maximum rate of increase (r_m) to the proportion of the population harvested annually. If the proportion harvested annually exceeds $r_m/2$ for a population at $K/2$, where K is the average size of the unharvested population (Caughley and Sinclair 1994), then the maximum sustained loss of the population is exceeded and the population will decline.

A third approach is to assess trends in parameters of the harvest and the harvested animals as indicators of the sustainability of the harvest. This approach provides an alternative when many population parameters are not available. Some indices that can be monitored are size and age distributions of harvested animals and capture per unit effort (CPUE). Although less desirable than monitoring population size directly, estimating and monitoring parameters of the harvest over time is the only approach for species whose populations are logistically difficult to sample, such as, sea turtles and seahorses (Vincent 1996).

Even when demographic parameters can be estimated, confidence in assessing sustainable take is often low due to unreliable estimates and lack of a thorough understanding of population dynamics. For example, Marsh (1997) evaluated the sustainability of a dugong fishery and although she estimated population parameters from many years of data collection, she was still unable to determine the sustainability of the fishery due to lack of essential data.

Biological Sustainability of the Marine Turtle Fishery

The sustainability of the Nicaragua marine turtle fishery can not be evaluated at this time by using population parameters because most of these data are not available. The data necessary to do a thorough evaluation of the biological sustainability of the fishery include harvest patterns, population estimates, and life history parameters. Only data on the fishery are available at this time, and therefore, an evaluation of harvest parameters and the harvested animals is the only approach available to provide some assessment of the biological sustainability of the fishery on the turtle populations. There are no published rigorous assessments on the biological sustainability of harvesting green turtles. I assessed the effect of the Miskitu Indian marine turtle fishery on green turtles and hawksbills by integrating my results on harvest rates, CPUE, and demographic parameters of harvested animals. The effect of the fishery on loggerhead and leatherback (*Dermochelys coriacea*) populations were not assessed because sufficient data from the fishery are not available at this time.

Methods

I used results from Chapters 2, 3, and 4; and comparisons of current harvest patterns with historical information about the fishery to assess the biological sustainability of the fishery. For green turtles, I categorized results as either not indicating overharvest or indicating overharvest for the following trend analyses: 1) the annual harvest rate for all sites combined and monthly harvest rates; 2) CPUE; and 3) size of harvested animals. For hawksbills, I review trends in monthly harvest levels and

compare them with historical harvest levels. No data are available on CPUE of hawksbills because their harvest is opportunistic during other activities, e.g., green turtle and lobster fisheries, and human presence on offshore cays and mainland nesting beaches. The evaluation of the impact of the fishery on Nicaragua marine turtle populations is preliminary because of the relatively short time period for which data are available and because of the indirect nature of available data.

Results

Green Turtles

Indicators that do not suggest overharvesting

From 1994 to 1996 the number of turtles killed/yr has remained relatively constant at approximately 10,000 to 11,000, based on landings at eight data collection sites. The trend in monthly harvest levels has remained relatively constant for all RAAS communities combined over a 2.5-yr period ($P = 0.29$), and for one RAAS community (Sandy Bay Sirpi) over a 6-yr period ($P = 0.78$). There has been no significant change in the net capture/unit effort (N-CPUE) in the RAAN over 13 mo ($P = 0.36$), in the RAAS over 17 mo ($P = 0.85$), or in one RAAS community (Sandy Bay Sirpi) for 48 mo of a 72-mo period ($P = 0.22$). No change in either N-CPUE or in harvest levels suggests that capture effort has not increased and that population levels have not, as yet, changed to the extent that they alter CPUE. For turtles landed in the RAAN at Puerto Cabezas (PC), Awastara (AW), and Sandy Bay (SB), the change in mean plastron lengths of ≤ 0.5 cm

for the 3.25-yr period was not significant ($P \geq 0.12$ for the three sites; $n = 3,886$ for PC; $n = 1,491$ for AW; and $n = 2,088$ for SB).

Indicators that suggest overharvesting

For turtles landed at four data collection sites in the RAAS, the mean plastron length decreased significantly by 4.6 cm over the 2.5-yr period ($P < 0.0001$, $n = 8,371$). Mean body mass of turtles ($n = 1,438$) landed in Puerto Cabezas in 1992/1993 was 10 kg less than turtles landed at one Miskitu village in 1968 (Nietschmann 1972, 1973; sample size not provided), a period prior to the Nicaragua processing plants. In recent years, in both the RAAN and RAAS, some turtlers reported decreasing the mesh size of their nets in order to catch smaller animals (V. Renales pers. com.; P. Julias pers. com.).

Hawksbill Turtles

Monthly harvest levels of hawksbills appear to have remained relatively constant from 1993 to 1996. Compared with the early 1970s (Nietschmann 1972, 1973), however, there has been a 479% decrease in the harvest rate of hawksbills by one Miskitu village.

Discussion

The data available, to date, on the fishery and harvested animals do not indicate conclusively whether or not the green turtle population is overharvested. Trend analyses on the size of harvested animals in the RAAN, N-CPUE, and monthly harvest rate in the RAAS do not indicate at this time that the foraging population is in decline. All analyses, however, were conducted over very short time periods (from 13 mo for the trend in

plastron length and CPUE to 72 mo for monthly harvest rates for one community in the RAAS), probably too short to detect changes in the population if they occurred. The significant decrease in mean plastron length of animals harvested in the RAAS during this study, however, is similar to one of several observations made twenty years ago, when harvest levels on the foraging ground were similar to current levels. At that time, researchers reported a decrease in the capture of larger animals, as well as a decrease in CPUE and a decrease in the nesting density of females at the Tortuguero rookery (Nietschmann 1972,1973, 1976, 1979a; Weiss 1976; Carr pers. com. to Nietschmann 1976).

The difference in results of the trend analyses for plastron length in the RAAN and RAAS is possibly due to differences in immigration rates of animals in the north and south regions compared to their respective harvest rates. The decrease in size of harvested animals in the RAAS could indicate that animals are immigrating onto the south foraging ground at a lower rate than they are harvested. The use of CPUE as an estimate or indicator of abundance can be problematic due to the relationship of CPUE and resource abundance (Hilborn and Walters 1992). In fisheries where search is highly efficient, the most effort is concentrated in areas where fish are abundant, resulting in hyperstability, a condition where CPUE remains high as fish abundance declines (Hilborn and Walters 1992). Thus, CPUE for the turtle harvest could remain unchanged while the turtle population is declining. At this time, the trend analysis on plastron length may be the best indication of the fisheries impact on the turtle population. The change in size in

the RAAS particularly over a relatively short time period suggests that overharvest is occurring.

Turtle Life History Traits and their Implications for Exploitation

In addition to the decrease in size of harvested animals in the RAAS, which suggests overharvesting, the life history traits of the species also suggest they could easily be overharvested. Sea turtles are long-lived, large-bodied marine vertebrates. They are slow to reach sexual maturity, surviving many years before reproducing. They lay several clutches within a season and many clutches throughout their reproductive life which is necessitated by high mortality of eggs and hatchlings. Thus, high survival of juveniles and adults is necessary to maintain stable populations (Congdon and Dunham 1994). The combination of these life history traits limits the ability of sea turtle populations to respond to chronic increases in juvenile or adult mortality through human-induced mortality, such as harvesting (Congdon et al. 1993, 1994; Congdon and Dunham 1994). Population modeling of loggerheads (Crouse et al. 1987; Crowder et al. 1994; Heppell et al. 1996a), hawksbills (Heppell et al. 1995), Kemp's ridleys, *Lepidochelys kempi*, (Heppell et al. 1996b), and several freshwater turtles species (*Kinosternon flavescens*, Heppell et al. 1996b; *Emydoidea blandingii*, Congdon et al. 1993; *Chelydra serpentina*, Congdon et al. 1994) has shown that survival of juveniles and adults must remain high to maintain stable populations of long-lived, slow-maturing, iteroparous species.

The sizes of green and hawksbill turtles harvested in the Nicaragua fishery indicate that they are mostly large juveniles or adults (this study). This conclusion is based on the relationship between the sizes of the animals harvested and the minimum sizes recorded for nesting females at the primary rookery Tortuguero, Costa Rica (see Chapter 3), the smallest males observed with sperm, and the smallest females observed with corpora lutea (for green turtles only; see Chapter 4). If the conclusions drawn from studies on population modeling for long-lived, slow maturing species, as summarized above, are valid for green and hawksbill turtles in Nicaragua, then the greatest pressure from the Miskitu Indian marine turtle fishery is on the life stages that are least able to withstand the effects of exploitation.

Green Turtles

Current regional perspective

In addition to the harvest of turtles in Nicaragua, an unknown number of green turtles and their eggs are also annually harvested throughout the greater Caribbean (Bacon 1975; Carr et al. 1982; Bacon et al. 1983; Meylan 1983; Pritchard and Trebbau 1984; Berry 1989; Eckert and Honebrink 1992; Eckert et al. 1992; Fuller et al. 1992; Rueda-Almonacid et al. 1992; Sybesma 1992; Scott and Horrocks 1993). Current annual regional harvest levels are unknown, but in Costa Rica alone the annual legal quota of green turtles has been 1,800 animals since 1983 (Ogren 1989; WWF 1997). An additional 1,780 females were estimated to be illegally harvested from the nesting beach at Tortuguero in 1997 (Troëng 1997). Because populations are highly migratory the

activities in one nation can adversely affect the resources of others. The additional harvests of green turtles throughout the Caribbean must be considered when assessing the impact of the Nicaragua green turtle fishery. Although the Nicaragua fishery is probably the largest in the region, this study only provides a partial view of the full magnitude of marine turtle exploitation that occurs in the Caribbean.

My data do not indicate conclusively that the Nicaragua green turtle foraging population is overharvested. However, based on the magnitude of the Miskitu Indian marine turtle fishery, as well as, the focus of the fishery on large juveniles and adults, there is clearly cause for concern. If current harvest levels are not sustainable, population declines probably will not be immediately apparent because of the species life history traits and fluctuations in harvest pressure. Animals with delayed sexual maturity have populations comprising a large percentage of animals in the juvenile stages which provides a temporary buffer against extinction (Bjorndal 1985). Fluctuations in harvest pressure can allow the population some level of recovery which would also make it unlikely that overharvest would be immediately apparent.

Historical events and their effect on the relative abundance of green turtle life stages

Local, regional, and international events can affect exploitation pressure on natural resources. International and local demand for green turtles for the past 500 years, has greatly affected population levels in the Caribbean (Carr 1954; Parsons 1962; Bjorndal 1980b, 1985; Bowen and Avise 1995; Jackson 1997). Bowen and Avise (1995) estimated a 99% decline in the Caribbean green turtle population since the late 1400s from an estimated 50 million adults, although the basis for their estimate is not given.

Jackson (1997) estimated the pre-Columbian adult green turtle population in the Caribbean at 600 million. Although estimates of the size of the pre-Columbian green turtle population in the Caribbean differ, clearly their numbers have declined drastically. Green turtle population declines in Bermuda, the Bahamas, the Cayman Islands, Jamaica, and various other localities throughout the greater Caribbean have been well documented (Lewis 1940; Carr 1954; Parsons 1962; King 1982; Dr. Archie Carr (interview) 1984). In order to avoid the shifting baseline syndrome (Pauly 1995; Sheppard 1995), the current harvest of green turtles from the Nicaragua foraging ground must be viewed from the perspective of an already depleted resource. The shifting baseline syndrome occurs when each new generation of scientists accepts the size of a population or species composition in an area at the start of their careers as the baseline and all subsequent evaluations are based on this starting point. If population size declines or species composition changes before the next generation of scientists begin their careers then the result is a gradual shift in the baseline and an assessment of species status using inappropriate reference points.

The following scenario is presented to explain what impact events in the 1900s have had on the relative abundance of four green turtle life stages and to explain the apparently large numbers of animals available to the Nicaragua fishery today. These life stages are: egg/hatchling; small juvenile; large juvenile; and adult. The relative abundance of each life stage is categorized qualitatively as “few”, “some”, “many”, and “lots”. By the turn of the 20th century, green turtle populations in the Caribbean had already been depleted by harvesting (Carr 1954; Parsons 1962; King 1982). Regional events since the beginning of the 1900s have resulted in variable exploitation pressure on

Nicaragua's green turtle foraging population affecting the relative abundance of animals in each life stage. Below I describe the probable effects these regional events might have had on the relative abundance of animals beginning in the 1900s when harvest levels are better documented.

From the early 1900s to 1968, approximately 2,000 to 4,000 green turtles were harvested annually from Nicaraguan waters and exported by Cayman turtlers (Ingle and Smith 1949; Parsons 1962) . In addition, an unknown quantity of turtles were harvested by Miskitu Indians and consumed locally. Although not well documented, total annual harvest levels in Nicaragua probably did not exceed 5,000 animals. Harvest at the Tortuguero, Costa Rica rookery, during the latter part of this 1900 to 1968 time period, included eggs, an estimated one-third of all females that emerged on to the beach, and an undetermined number of both males and females from nearshore waters adjacent to the nesting beach (Carr 1954, 1969; Carr et al. 1978). Estimates of harvest levels from the Tortuguero rookery from 1900 to 1968 are not available. Because juvenile and adult life stages are long (i.e, 20 to 50 yrs for juveniles and at least 30 yrs for adults) compared to the egg/hatchling stage (i.e., 60 days to 1 yr), animals accumulate in the juvenile and adult life stages. Because adults were probably the focus of the fishery and exploitation levels relatively low and variable between 1900 and 1968, the relative abundance of each life stage by 1968 was probably "some" production in the egg/hatchling, "some" small and "many" large juveniles, and "some" adults (Figure 5.1). The relative abundance of large juveniles would decline more slowly than the other stages because of continued but variable recruitment from the previous stage.

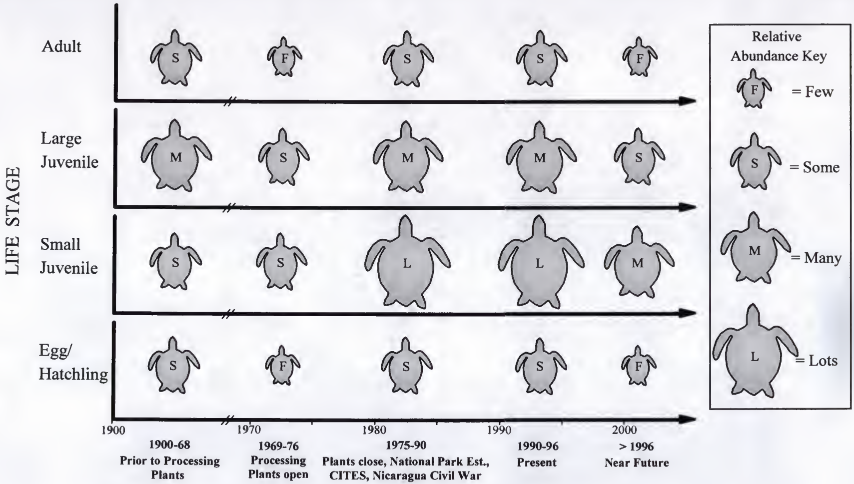


Figure 5.1. Importance of regional historical events on the relative abundance of green turtles, *Chelonia mydas*, by life stage on the Nicaragua foraging ground. For each period, the relative abundance in a life stage reflects the impact of the event on that life stage by the end of the period in relation to the size of the same stage and earlier stage during the previous event. For example, the increased harvest pressure that resulted from the operation of turtle processing plants from 1969-76 would have decreased the relative abundance of the large juvenile and adult stages resulting in decreased egg/hatchling relative abundance.

In early 1969, the first of three marine turtle processing plants on Nicaragua's coast began operations. From 1969 to 1976, an estimated 6,000 to 15,000 or more green turtles were harvested annually from the western Caribbean (Nietschmann 1973, 1979b; Carr et al. 1978). Due to the increase in exploitation during this period, by the end of the period, the relative abundance of the adult life stage probably had declined to a "few" animals and as a result there would be relatively "few" egg/hatchlings produced. Because small juveniles were not targeted in the fishery their relative abundance would have probably remained at "some" animals. Due to the increased demand for turtles during this period and declining abundance in the adult stage, harvest pressure probably increased in the large juvenile stage as reflected in fewer adults captured (Nietschmann 1972, 1973, 1976). Thus, the relative abundance in the large juvenile stage would have declined to "some" (Figure 5.1).

Events that occurred during the next period, 1975-1990, afforded animals and eggs at the Tortuguero rookery and animals at the Nicaragua foraging ground some degree of protection. By the mid-1970s, the beach at Tortuguero was declared a national park. By 1977, all turtle processing plants were closed in Nicaragua and the country became a signatory of CITES (Convention on International Trade in Endangered Species of Wild Fauna and Flora; Hemley 1994). In the 1980s, Nicaragua was involved in a decade-long civil war. Key informants from both the RAAN and RAAS reported a decrease in turtling activity during the war. The occurrence of these events from 1975 to 1990 would have decreased harvest pressure on all life stages, so that by the end of the

period all life stages would have had approximately 15 years to increase, thus, resulting in a pulse of animals into the population at all life stages (Figure 5.1).

Since the end of the war, harvest pressure has increased to current levels of approximately 10,000 - 11,000 animals/yr. Although results from evaluating the impact of the fishery on the foraging population during the 1990s are inconclusive there is a risk of evoking the shifting baseline syndrome due to a lack of quantitative data prior to this study and its short duration. However, based on the data obtained during this study, there is little evidence to indicate that the relative abundance of animals has declined in any of the life stages since 1990, the end of the civil-war (Figure 5.1).

Although the Caribbean green turtle population is still greatly reduced since pre-Columbian times, the increase in the abundance of animals from the mid-1970s to 1990 was apparently sufficient to enable the harvest to reach current levels. If demand continues, harvest levels will probably remain high for several more years as small juveniles in the "pipeline" move into the harvested size class. The relative abundance of the adult stage will decrease more rapidly than the large juvenile stage because large juveniles are also harvested in the fishery and thus recruitment into the adult stage will decrease. In contrast, recruitment into the large juvenile stage will continue for a longer period than into the adult stage because small juveniles are generally not harvested in the fishery (Figure 5.1). If current annual harvest levels of large juveniles and adults are greater than annual recruitment into these life stages, then eventually the fishery will show signs of a declining population by a decrease in CPUE or in the size of animals harvested.

Hawksbill Turtles

Historical information indicates that hawksbill populations are severely depleted in Nicaragua (Nietschmann 1981) and throughout the greater Caribbean region (Parsons 1972; King 1982; Bjorndal et al. 1993; Meylan 1997a). Severely depleted (small) populations are more vulnerable to extinction primarily from environmental and demographic stochastic effects (Caughley and Sinclair 1994; Caughley and Gunn 1996) and thus these populations should not be harvested.

Conclusions

My data do not show conclusively whether or not the current harvest level of green turtles is sustainable. Nor do they indicate how many animals of each size class and sex can be sustainably harvested. However, there is strong evidence that the life stages (juvenile and adult) that affect population change the most (Crouse et al. 1987; Congdon et al. 1993, 1994; Congdon and Dunham 1994; Crowder et al. 1994; Heppell et al. 1995; Heppell et al. 1996a, b) are most heavily impacted by the fishery. There are also some signs of overharvesting (this study). Because the larger (i.e., older) animals are targeted, the population will be slow to recover from a decline, if overharvesting is occurring. Therefore, managing the fishery is critical. Clearly, more data are needed to assess the magnitude of the Nicaragua Caribbean fishery and to thoroughly evaluate its impact on populations of green, hawksbill, and loggerhead turtles. The Miskitu Indian marine turtle fishery should continue to be monitored and future efforts to characterize the populations and model population dynamics are needed to assess the effects of

different harvest levels. This information will help ensure the long-term survival of these endangered species, both as an important cultural and economic resource to coastal inhabitants of Nicaragua and as a valuable natural resource to people throughout the greater Caribbean (see Chapter 6 for recommendations).

CHAPTER 6
MANAGEMENT IMPLICATIONS, RECOMMENDATIONS, AND RESEARCH NEEDS

Implications for Managing the Marine Turtle Fishery

Today, marine turtles are still an important resource to coastal inhabitants of Caribbean Nicaragua. Turtles are an inexpensive source of meat and a source of income with which to purchase other goods and services. As with most biological resources, however, marine turtle populations can not withstand unlimited exploitation levels. Results from this study suggest that current harvest patterns could be exceeding the ability of marine turtle populations to maintain themselves. Indications of overharvest are the decrease in mean size and weight of harvested green turtles (*Chelonia mydas*) and a decrease in mesh size used in turtlers' nets. In addition, modeling of long-lived, slow-maturing iteroparous species, such as sea turtles, has demonstrated that the juvenile and adult life stages are the most sensitive to population growth or decline. Thus, the fact that the Nicaragua marine turtle fishery targets large juveniles and adults is cause for concern.

There is a need to regulate the number, size, and sex of animals harvested. Because sea turtles are long-lived and slow to reach sexual maturity, low survival in the early life stages limits their ability to respond to increased mortality in later life stages (Congdon et al. 1993, 1994; Congdon and Dunham 1994), which adds to the challenge of

managing resource use. As with any species, overexploitation can only be avoided if harvest rates do not exceed recruitment rates into the portion of the population harvested. Because sea turtle life history traits restrict their ability to respond to persistent and high levels of overharvest, humans must control and carefully manage their patterns of exploitation.

Nutritional and economic needs and desires of coastal inhabitants should be met with a variety of locally available resources and economic opportunities. The economic base of people living on the Caribbean coast of Nicaragua, however, needs to be broadened by developing alternative economic resources to mitigate pressure on sea turtles. Unless properly managed, the Miskitu Indian marine turtle fishery is likely to negatively impact sea turtle populations throughout the greater Caribbean, and eventually affect the availability of marine turtles as a resource for future generations of Miskitu Indians.

Governmental institutions usually are mandated to establish policies for natural resource use, develop management plans, and enforce laws that govern resource use. However, governmental institutions are often not able, neither financially, technically, or politically, to manage and enforce natural resource use. For these reasons, governmental institutions must involve local resource users in the decision-making process and implementation of managing resource use (Bromley and Cernea 1989; Rettig et al. 1989; Ostrom 1990; Pomeroy 1995; Harrington and Gallucci 1996; Jain 1996; Stocks 1996). In Nicaragua, the government institution, Ministerio del Ambiente y Recursos Naturales (MARENA), mandated to manage natural resources in the country is constrained by

insufficient finances, lack of technical expertise, and little political influence on the Caribbean coast. In addition, they are also separated geographically and culturally from marine turtle harvesters and consumers on that coast. Thus, the marine turtle fishery needs to be co-managed by all the stakeholders of this resource, which includes but is not limited to the turtling communities, marine turtle butchers, MARENA, regional autonomous governments (RAAN and RAAS), coastal municipalities, navy, and nongovernmental organizations (NGO). Involving resource users and other interest groups in co-management of resource exploitation improves the chances of successfully achieving a sustainable harvest and enforcing regulations to conserve the resource (Pomeroy 1995; Harrington and Gallucci 1996; Jain 1996; also see Pinkerton 1989 for examples of various co-management schemes). One major difference between the Nicaragua marine turtle fishery and other fisheries is that in Nicaragua there is only one user group (coastal indigenous people), and therefore, no competition for the resource occurs between the different stakeholders as with most other fisheries. Thus, co-management of the marine turtle fishery in Nicaragua is needed to regulate the fishery for the sustainable use of the resource, but not to regulate resource allocation, and spatial or temporal access to the resource among user groups.

In Nicaragua, establishing collaborative agreements among the various stakeholders on the Caribbean coast will be particularly challenging because of past events. One result from the negotiations to end the Nicaragua civil war of the 1980s was an agreement by the central government to grant autonomy to the people living in the eastern Caribbean lowlands of the country. Two regional autonomous governments (one

in the north and one in the south) were established to govern regional issues, including the use and management of natural resources. In addition, by legislative decree, the inhabitants of the Caribbean lowlands were given the right to use and enjoy the natural resources of the autonomous regions (D. Castro pers. com.). This clearly presents a conflict between the establishment of regulations by the central government and resource users of the autonomous regions.

Another event was the establishment of the Reserva de Biósfera de las Comunidades Indígenas y Cayos Miskitos (Indigenous Communities and Miskito Cays Biosphere Reserve) in 1991. Impetus for the reserve was the concern for overuse of and competition for the natural resources in the region. The reserve encompasses 24,000 km^2 in the RAAN, including 38 indigenous communities, and offshore cays and reefs (Equipo Técnico de Planificación 1995). A draft management plan for the reserve has been written based on the concerns of the indigenous communities for competition with non-Miskitu groups for natural resources (D. Castro pers. com.). However, it does not include the management of the marine turtle fishery because, again, there is no competition from non-Miskitu interests for the harvest of marine turtles. There is still no formal institutional structure for management of the reserve, and as a result, development of the reserve and activities are essentially paralyzed (Jain 1996). Although marine turtles were not considered in the draft management plan for the reserve, an important outcome in the process of writing the draft plan was the inclusion of the indigenous communities. In addition, the fact that the communities do not compete with other interest groups for

marine turtles makes the issue less political and places the responsibility of managing the turtle harvest squarely in their hands.

The people that most directly depend on the resource will either prosper in the long-term when exploitation levels are managed within biological limits of the resource or suffer the consequences when the resource is overharvested and no longer available for human use. Therefore, the ultimate burden of maintaining a sustainable marine turtle harvest lies with the people that depend on the resource and are the direct beneficiaries of sound resource use. This is not to imply that government institutions, NGOs, and the scientific community do not have a role to play in assisting resource users to develop more sustainable resource use practices. Through associations with conservation organizations and government institutions, resource users should be supported and assisted in the development of alternative resources and economic options, provided with knowledge and information about the resource, and assisted in obtaining data to make the most informed management decisions (Stocks 1996). Although, approaches to developing collaborative agreements with local resource users and finding solutions to conservation issues can sometimes be transferred among geographic locations, human use patterns of natural resources are often site specific and biological constraints are species specific and often require unique solutions to conservation issues.

The conservation and management of the marine turtle fishery on the Caribbean coast of Nicaragua needs to begin within the turtling communities. It is absolutely imperative that the Miskitu people, as a community, be actively involved in the decision making process of establishing the rules and regulations needed to manage this fishery for

a more sustained harvest. Although community members are the harvesters of marine turtles and the beneficiaries of the resource, they are the preferred stewards because they have the most to lose if the resource declines. Since currently no marine turtle fishery management plan exists, it is important that the turtling communities agree to establish regulations that will, at the very least, prevent the harvest from increasing. Simultaneously, education and training programs should be implemented to provide coastal inhabitants with biological knowledge about the species and training in collecting and analyzing biological data on the turtles, as well as, sociological and economic data about the marine turtle fishery.

Because the cost of marine turtle meat is currently the second, only to fish, least expensive source of protein on the Caribbean coast, alternative sources of inexpensive protein need to be explored and additional economic opportunities developed. In exchange, turtlers will need to agree to restrict their harvest of marine turtles. Below I have listed the most immediate management actions that I recommend be considered, modified if necessary, and implemented by the turtling communities. The section on management recommendations is followed by a section on research needs. I list the most important research needs that will provide us with the additional information needed to better manage the marine turtle fishery for the long-term survival of turtle populations.

Management Recommendations

The following recommendations are made with the limited data available and will need revision when data on the turtle populations in the region have been obtained.

Regulations that impinge on social, economic, and cultural aspects of the turtlers, turtle butchers, and coastal inhabitants will need to be discussed and agreed on among the various interest groups.

1. Continue monitoring the marine turtle harvest through local community data collectors. For each turtling trip at least the following information should be recorded: turtler's community of residence, trip dates, number of days turtling, capture location(s), harvest method, number of nets (if used), number of turtlers, number of turtles captured by species, and size and sex of animals captured.
2. Expand monitoring of the harvest to include the communities of Rama Indians that harvest turtles, as well as, the additional three Miskitu Indian communities in the RAAN.
3. Establish monthly harvest quotas for each turtling community based on current harvest rates to prevent the total annual harvest of green turtles from exceeding 10,000 animals. This could be accomplished by monitoring monthly harvest rates for each community and adjusting harvest levels in subsequent months.
4. Establish restrictions on the harvest of green turtles to lessen the impact of the harvest on the population. Because insufficient data are available to determine which life stage (i.e., juveniles or adults) should be protected to maximize population growth rates, the following discussion of advantages and disadvantages of various strategies is provided. However, for any strategy to be effective, it is imperative that the Caribbean Nicaragua turtlers are included in discussions about restricting the harvest and deciding which restrictions to adopt.

a. Decrease the number of female adult and large juvenile green turtles harvested. Even though the sex ratio indicates that more females occur in the foraging population, females are more valuable for population growth and, therefore are important to obtaining the highest population growth rate.

b. Establish a closed season on the harvest of females > 88.0 cm carapace length (CL) from January through May to allow the migration of reproductively active females of the year to migrate to the nesting beaches. However, partial year-around closed seasons for specific sexes or size classes are less protective and difficult to enforce. They are also potentially confusing and are more easily misinterpreted.

c. Protect male and female green turtles < 88.0 cm CL throughout the year. The presence of corpora lutea indicates that approximately 50% of females > 88.0 cm CL are reproductively mature (see Figure 4.5B). The protection of animals < 88.0 cm CL will allow some animals to reach sexual maturity and reproduce. Based on modeling studies of long-lived, iteroparous species increased survival in the juvenile life stage results in the highest population growth rate. If only immature animals are protected, however, the increased harvest pressure on mature animals could result in fewer reproductive individuals in the population.

d. Protect male and female green turtles ≥ 88.0 cm CL throughout the year. This would provide protection to the majority of animals that have already reached sexual maturity. If only mature animals are protected, however, the increased

harvest pressure on juvenile animals could eventually result in a smaller adult population because fewer juveniles would recruit into the adult stage to replace adults lost to reproductive senescence and mortality outside the Nicaragua turtle fishery.

6. Prohibit the transportation of green turtles to the Río Coco region (located on the border with Honduras) and to interior savannah communities. Consumption of turtle meat in these areas is not yet an established custom and should be discouraged to contain the market for sea turtle products.
7. Prohibit the sale and use of any marine turtle species for bait in other fisheries.
8. Establish a minimum mesh size of 16 in (41 cm) bar for turtle entanglement nets.
9. Establish a country-wide ban on the harvest, use, and sale of hawksbill (*Eretmochelys imbricata*) meat, eggs, and tortoiseshell.
10. Promote the live release of loggerheads (*Caretta caretta*) and leatherbacks (*Dermochelys coriacea*) when they are captured.
11. Promote the cultivation and consumption of alternative, inexpensive sources of animal protein to be bred and sold by turtlers to substitute lost income from the sale of marine turtles. Alternative meat sources will need to be evaluated for cultural acceptance and economic viability.
12. Develop and promote additional economic opportunities, such as, tourism, the sale of goods produced by the woman's sewing club, animal husbandry, and the production and sale of fruit currently produced in the communities.

13. Develop alternative economic opportunities for the turtle butchers in Puerto Cabezas to decrease their dependence on marine turtles for income.
14. Identify and establish marine turtle capture sites to be used for long-term monitoring of population trends.
15. Develop alternative sources of material, instead of coral, to use in the footline of turtle nets and encourage turtlers to reuse already harvested coral. Encourage the use of dead coral that has been washed-up during storms and litters many beaches and shoreline areas.
16. Establish and enforce a law that requires the use of Turtle Excluder Devices (TEDs) on all shrimp trawlers to decrease the drowning or additional harvest of turtles in shrimp nets.

Recommendations for Future Research

1. Conduct in-water studies to characterize the foraging populations. Studies should include the following components:
    a) determine relative density, and survival and growth rates of turtles by conducting mark/recapture studies,
    b) determine characteristics (i.e., size, sex, and genetic stock) of juvenile populations,
    c) determine recruitment rate of turtles into the harvested populations,
    d) determine habitat use by size class and sex and determine available habitat, and
    e) determine migratory patterns to and from nesting beaches.

2. Determine the proportion of animals in various reproductive states (i.e., immature, mature, reproductively active) from a random sample of harvested animals and monitor changes over time.
3. Develop a population projection model to evaluate survival outlook of the foraging population based on current harvest levels and population density, and to evaluate different management strategies.
4. Survey hawksbill nesting populations along the southern Caribbean coast of the country to determine their status and harvest pressure.
5. Characterize the population of loggerheads in Nicaragua's Caribbean waters. Studies would include the following components:
    a) determine size and sex of animals captured in the fishery,
    b) determine relative density of turtles by conducting mark/recapture studies,
    c) determine size, sex, and genetic stock of the in-water population, and
    d) determine if the population on the foraging ground is resident, seasonal, or migratory.
6. Quantify the amount of coral harvested for use in the fishery and investigate impacts to coral communities.

APPENDIX A

MINIMUM NUMBER AND (PERCENT) OF MARINE TURTLES CAPTURED IN THE REGION AUTONOMA DEL ATLANTICO NORTE, NICARAGUA FOR THE PERIODS FEBRUARY 1994 TO JANUARY 1995 AND DECEMBER 1995 TO APRIL 1997

	Chelonia mydas					*Eretmochelys imbricata*				*Caretta caretta*			
Capture Location	AW[a]	DK[b]	SB[c]	UNK[d]	TOT[e]	AW	DK	SB	TOT	AW	DK	SB	TOT
Awastara	18 (0.3)	0 (0)	0 (0)	0 (0)	18 (0.2)	0 (0)	0 (0)	0 (0)	0 (0)	4 (0.7)	0 (0)	0 (0)	4 (0.6)
Broston Bar	93 (1.3)	0 (0)	123 (4.3)	0 (0)	216 (1.9)	0 (0)	0 (0)	1 (5.0)	1 (1.3)	2 (0.4)	0 (0)	5 (12.2)	7 (1.1)
Buhnitara	14 (0.2)	0 (0)	0 (0)	10 (3.8)	24 (0.2)	0 (0)	0 (0)	0 (0)	0 (0)	1 (0.2)	0 (0)	0 (0)	1 (0.2)
Corn Fish	0 (0)	0 (0)	22 (0.8)	0 (0)	22 (0.2)	0 (0)	0 (0)	1 (5.0)	1 (1.3)	0 (0)	0 (0)	0 (0)	0 (0)
D.D. Rock	10 (0.1)	87 (5.4)	0 (0)	0 (0)	97 (0.8)	0 (0)	1 (4.3)	0 (0)	1 (1.3)	0 (0)	0 (0)	0 (0)	0 (0)
Deadman Bar	0 (0)	0 (0)	56 (2.0)	0 (0)	56 (0.5)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)
Diamond Patch	69 (1.0)	10 (0.6)	1,157 (40.7)	0 (0)	1,236 (10.6)	0 (0)	0 (0)	7 (35.0)	7 (9.0)	2 (0.4)	0 (0)	5 (12.2)	7 (1.1)
Dos Palitos	0 (0)	0 (0)	28 (1.0)	0 (0)	28 (0.2)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)
Engine Bar	0 (0)	0 (0)	104 (3.7)	0 (0)	104 (0.9)	0 (0)	0 (0)	1 (5.0)	1 (1.3)	0 (0)	0 (0)	2 (4.9)	2 (0.3)
Franklin Reef	17 (0.2)	0 (0)	0 (0)	0 (0)	17 (0.1)	1 (2.9)	0 (0)	0 (0)	1 (1.3)	0 (0)	0 (0)	0 (0)	0 (0)
Inin	0 (0)	0 (0)	18 (0.6)	0 (0)	18 (0.2)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)
Karil	48 (0.7)	0 (0)	0 (0)	0 (0)	48 (0.4)	0 (0)	0 (0)	0 (0)	0 (0)	4 (0.7)	0 (0)	0 (0)	4 (0.6)
Leimarka	895 (13.0)	113 (7.0)	57 (2.0)	0 (0)	1,065 (9.2)	3 (8.6)	2 (8.7)	0 (0)	5 (6.4)	172 (31.9)	2 (3.9)	0 (0)	174 (27.6)
London Reef	0 (0)	3 (0.2)	22 (0.8)	0 (0)	25 (0.2)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)
Mackensie Bar	13 (0.2)	91 (5.6)	20 (0.7)	0 (0)	124 (1.1)	0 (0)	1 (4.3)	0 (0)	1 (1.3)	1 (0.2)	1 (2.0)	0 (0)	2 (0.3)

Capture Location	*Chelonia mydas*					*Eretmochelys imbricata*				*Caretta caretta*			
	AW[a]	DK[b]	SB[c]	UNK[d]	TOT[e]	AW	DK	SB	TOT	AW	DK	SB	TOT
Maras Cay	206 (3.0)	110 (6.8)	13 (0.5)	0 (0)	329 (2.8)	1 (2.9)	1 (4.3)	0 (0)	2 (2.6)	5 (0.9)	4 (7.8)	1 (2.4)	10 (1.6)
Miskito Cay	324 (4.7)	2 (0.1)	6 (0.2)	0 (0)	332 (2.9)	4 (11.4)	0 (0)	0 (0)	4 (5.1)	35 (6.5)	0 (0)	0 (0)	35 (5.5)
Bet. Miskito & Maras	43 (0.6)	0 (0)	4 (0.1)	0 (0)	47 (0.4)	0 (0)	0 (0)	0 (0)	0 (0)	2 (0.4)	0 (0)	2 (4.9)	4 (0.6)
Muna de Nasa	0 (0)	0 (0)	26 (0.9)	0 (0)	26 (0.2)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)
Nasa Cay	468 (6.8)	106 (6.6)	204 (7.2)	0 (0)	778 (6.7)	1 (2.9)	1 (4.3)	1 (5.0)	3 (3.8)	48 (8.9)	0 (0)	0 (0)	48 (7.6)
Nasa Sautka	0 (0)	0 (0)	17 (0.6)	0 (0)	17 (0.1)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	1 (2.4)	1 (0.2)
Ned Thomas	0 (0)	29 (1.8)	80 (2.8)	0 (0)	109 (0.9)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	2 (3.9)	2 (4.9)	4 (0.6)
Nee Reef	0 (0)	0 (0)	120 (4.2)	0 (0)	120 (1.0)	0 (0)	0 (0)	2 (10.0)	2 (2.6)	0 (0)	0 (0)	7 (17.1)	7 (1.1)
Oben Sol	0 (0)	15 (0.9)	0 (0)	0 (0)	15 (0.1)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)
Papta	22 (0.3)	12 (0.7)	0 (0)	0 (0)	34 (0.3)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)
Plaik	0 (0)	0 (0)	86 (3.0)	0 (0)	86 (0.7)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)
Ples Raya	403 (5.8)	7 (0.4)	95 (3.3)	0 (0)	505 (4.3)	6 (17.1)	0 (0)	0 (0)	6 (7.7)	11 (2.0)	0 (0)	5 (12.2)	16 (2.5)
Reef	0 (0)	0 (0)	87 (3.1)	0 (0)	87 (0.7)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	3 (7.3)	3 (0.5)
Siakualaya	21 (0.3)	65 (4.0)	0 (0)	0 (0)	86 (0.7)	0 (0)	3 (13.0)	0 (0)	3 (3.8)	1 (0.2)	9 (17.6)	0 (0)	10 (1.6)
Sleps	18 (0.3)	0 (0)	0 (0)	0 (0)	18 (0.2)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)
Southeast Rock	24 (0.3)	0 (0)	0 (0)	26 (9.9)	50 (0.4)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)
Tistan Reef	0 (0)	0 (0)	13 (0.5)	0 (0)	13 (0.1)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)
Udlock Rock	0 (0)	0 (0)	28 (1.0)	0 (0)	28 (0.2)	0 (0)	0 (0)	1 (5.0)	1 (1.3)	0 (0)	0 (0)	2 (4.9)	2 (0.3)
Uiplin	202 (2.9)	190 (11.8)	72 (2.5)	0 (0)	464 (4.0)	1 (2.9)	4 (17.4)	1 (5.0)	6 (7.7)	3 (0.6)	11 (21.6)	1 (2.4)	15 (2.4)
Waham	71 (1.0)	146 (9.0)	0 (0)	0 (0)	217 (1.9)	0 (0)	3 (13.0)	0 (0)	3 (3.8)	2 (0.4)	3 (5.9)	0 (0)	5 (0.8)
Waltara	0 (0)	2 (0.1)	0 (0)	0 (0)	2 (0.02)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	2 (3.9)	0 (0)	2 (0.3)
White Hole Cay	31 (0.4)	176 (10.9)	27 (0.9)	0 (0)	234 (2.0)	0 (0)	5 (21.7)	0 (0)	5 (6.4)	2 (0.4)	7 (13.7)	0 (0)	9 (1.4)

Capture Location	*Chelonia mydas*					*Eretmochelys imbricata*				*Caretta caretta*			
	AW[a]	DK[b]	SB[c]	UNK[d]	TOT[e]	AW	DK	SB	TOT	AW	DK	SB	TOT
Witties	3,862 (56.1)	340 (21.1)	284 (10.0)	216 (82.1)	4,702 (40.5)	18 (51.4)	0 (0)	3 (15.0)	21 (26.9)	244 (45.3)	5 (9.8)	3 (7.3)	252 (39.9)
Won kluna	0 (0)	0 (0)	0 (0)	11 (4.2)	11 (0.1)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)
Yankalaya	18 (0.3)	0 (0)	0 (0)	0 (0)	18 (0.2)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)
TOTAL	6,890	1,614	2,844	263	11,611	35	23	20	78	539	51	41	631

[a] AW = Awastara
[b] DK = Dakra
[c] SB = Sandy Bay
[d] UNK = Unknown community
[e] TOT = Total for the row by species.

APPENDIX B
MINIMUM NUMBER OF MARINE TURTLES CAPTURED BY COMMUNITY
IN THE REGION AUTONOMA DEL ATLANTICO SUR, NICARAGUA

Rio Grande Bar (April 1991 to December 1996, for 46 mo of the 69-mo period)

	Chelonia mydas		*Eretmochelys imbricata*		*Caretta caretta*	
Capture Location	Number	Percent	Number	Percent	Number	Percent
Big Criki	110	2.0	0	0	0	0
Big Northeast	63	1.1	0	0	0	0
Big Shoal	42	0.7	0	0	0	0
Boor Bila	49	0.9	0	0	0	0
Brisbane Shoal	98	1.7	0	0	0	0
De Tronco	347	6.2	2	16.7	4	40.0
Duban	423	7.5	0	0	0	0
East	82	1.5	1	8.3	0	0
East/Northeast	43	0.8	1	8.3	1	10.0
East/Southeast	18	0.3	1	8.3	1	10.0
For Now This	15	0.3	0	0	0	0
Half-way Bank	1,164	20.7	3	25.0	1	10.0
Jaco Shoal	135	2.4	0	0	0	0
Jorge Hall	32	0.6	0	0	0	0
Karmutra Bank	787	14.0	1	8.3	1	10.0
Kuala Cay	45	0.8	0	0	0	0
Northeast	260	4.6	1	8.3	2	20.0
North/Northeast	353	6.3	2	16.7	0	0
Northwest	4	0.1	0	0	0	0

Capture Location	Chelonia mydas		Eretmochelys imbricata		Caretta caretta	
	Number	Percent	Number	Percent	Number	Percent
Southeast	228	4.1	0	0	0	0
Vietnam Bank	1,076	19.1	0	0	0	0
Vietnam Sur	30	0.5	0	0	0	0
Washer Woman	195	3.5	0	0	0	0
Yankee Shoal	25	0.4	0	0	0	0
TOTAL	5,624	100.1	12	99.9	10	100

Data for 1991 to 1993 were provided by the Centro de Investigaciones y Documentación de la Costa Atlántica.

Sandy Bay Sirpi (January 1991 to December 1996, for 64 mo of the 72-mo period)

Capture Location	Chelonia mydas		Eretmochelys imbricata		Caretta caretta	
	Number	Percent	Number	Percent	Number	Percent
Alge Bank	9	0.2	0	0	0	0
Auhya Pihni	376	7.6	1	2.8	0	0
Big Shoal	39	0.8	0	0	0	0
Buhni Pahni	125	2.5	0	0	1	3.6
Clar Cay	198	4.0	1	2.8	0	0
Devil Hole	315	6.4	0	0	1	3.6
Diamond Spot	145	2.9	0	0	6	21.4
Diapapa Bank	4	0.1	0	0	0	0
Dos Hermanos	28	0.6	0	0	0	0
Family Shoal	392	7.9	2	5.6	12	42.9
Half-way Bank	470	9.5	6	16.7	0	0
Hawksbill Bank	141	2.9	14	38.9	0	0
Jim Time	156	3.2	1	2.8	1	3.6
Kiski Tara	30	0.6	0	0	0	0
Liwa Pin	90	1.8	0	0	0	0

Capture Location	*Chelonia mydas* Number	*Chelonia mydas* Percent	*Eretmochelys imbricata* Number	*Eretmochelys imbricata* Percent	*Caretta caretta* Number	*Caretta caretta* Percent
Lousigsa	133	2.7	6	16.7	3	10.7
Mabra Pin Lalma	49	1.0	0	0	0	0
Man O' War Cay	128	2.6	0	0	0	0
Man O' War Lalma	17	0.3	0	0	0	0
Man O' War Waupasa	43	0.9	0	0	0	0
Masmaslaya	71	1.4	0	0	0	0
Nicro Sal Bank	41	0.8	1	2.8	0	0
North Schooner	108	2.2	0	0	0	0
Rolan Tara	31	0.6	0	0	0	0
Salikan Bila	6	0.1	0	0	0	0
Sandy Bay Sirpi	25	0.5	0	0	0	0
Skuna Yahbraro	13	0.3	0	0	0	0
Snukrik	13	0.3	0	0	0	0
South Schooner	606	12.3	0	0	0	0
Tingni Tara	108	2.2	0	0	0	0
Wainwin	633	12.8	4	11.1	1	3.6
Wainwin Munnita	17	0.3	0	0	0	0
Wainwin Waupasa	6	0.1	0	0	0	0
Wanklua	350	7.1	0	0	3	10.7
TOTAL	4,937	99.9	36	100.2	28	100.1

Data for 1991 to 1993 were provided by the Centro de Investigaciones y Documentación de la Costa Atlántica.

Set Net (July 1994 to December 1996, 30 mo)

	Chelonia mydas		*Eretmochelys imbricata*		*Caretta caretta*	
Capture Location	Number	Percent	Number	Percent	Number	Percent
Columbilla Cay	9	1.2	0	0	0	0
Compass	61	8.0	2	11.1	1	5.0
Crowning Spot	22	2.9	1	5.6	0	0
Fowlshit Bank	488	64.2	4	22.2	9	45.0
Long Reef	149	19.6	9	50.0	10	50.0
Maroon Cay	22	2.9	2	11.1	0	0
North Long Reef	9	1.2	0	0	0	0
TOTAL	760	100	18	100	20	100

Tasbapaune (November 1993 to December 1996, for 36 mo of the 38-mo period)

	Chelonia mydas		*Eretmochelys imbricata*		*Caretta caretta*	
Capture Location	Number	Percent	Number	Percent	Number	Percent
Au Dakra	12	0.2	0	0	0	0
Buscan	902	14.7	12	17.6	17	23.9
Cama Cay	26	0.4	1	1.5	1	1.4
Columbilla Cay	10	0.2	0	0	0	0
Haulover	1,525	24.9	20	29.4	0	0
Joe Bush	131	2.1	0	0	1	1.4
Kiama Cay	22	0.4	0	0	0	0
Kings Cay	98	1.6	1	1.5	1	1.4
Karaslaya	33	0.5	0	0	0	0
Kingsman Bank	426	6.9	5	7.4	0	0
Middle Bank	815	13.3	8	11.8	23	32.4
Prata Shoal	489	8.0	4	5.9	22	31.0

	Chelonia mydas		*Eretmochelys imbricata*		*Caretta caretta*	
Capture Location	Number	Percent	Number	Percent	Number	Percent
Patan	496	8.1	6	8.8	0	0
Rivas	1,150	18.7	11	16.2	6	8.5
TOTAL	6,135	100	68	100.1	71	100

APPENDIX C

MINIMUM NUMBER OF GREEN TURTLES, *CHELONIA MYDAS*, LANDED AT EACH SITE ON THE CARIBBEAN COAST OF NICARAGUA FROM 1991 TO 1996

Región Autónoma del Atlántico Norte	1991[a]		1992[b]		1993[c]	
	RECORDED (months)	ESTIMATED TOTAL	RECORDED (months)	ESTIMATED TOTAL	RECORDED (months)	ESTIMATED TOTAL
Awastara	N/A	N/A	N/A	N/A	N/A	N/A
Dakra	N/A	N/A	N/A	N/A	N/A	N/A
Sandy Bay	N/A	N/A	1,238 (8)	1,857[e]	1,694 (12)	1,694
Puerto Cabezas	833 (4)	2,497[e]	1,461 (9)	1,947[e]	1,143 (5)	2,746[e]
Subtotal	833	2,497[e]	2,699	3,804[e]	2,837	4,440[e]
Región Autónoma del Atlántico Sur						
Río Grande Bar	757 (3)	3,025[e]	1,475 (4)	4,427[e]	N/A	N/A
Sandy Bay Sirpi	377 (7)	647[e]	1,007 (10)	1,209[e]	838 (7)	1,438[e]
Set Net	N/A	N/A	N/A	N/A	N/A	N/A
Tasbapaune	N/A	N/A	N/A	N/A	336 (2)	2,016[e]
Subtotal	1,134	3,672[e]	2,482	5,636[e]	1,174	3,454[e]
TOTAL	1,967	6,169[e]	5,181	9,440[e]	4,011	7,894[e]

Región Autónoma del Atlántico Norte	1994[d]		1995[d]		1996[d]	
	RECORDED	ESTIMATED TOTAL	RECORDED	ESTIMATED TOTAL	RECORDED	ESTIMATED TOTAL
Awastara	335 (11)	365[e]	22 (2)	302[f]	731 (12)	731
Dakra	637 (11)	695[e]	62 (2)	422[f]	359 (12)	359
Sandy Bay	1,269 (11)	1,384[e]	349 (3)	1,312[f]	1,573 (12)	1,573
Puerto Cabezas	3,579 (12)	3,579	2,987 (12)	2,987	2,530 (12)	2,530
Subtotal	5,820	6,023[e]	3,420	5,023[f]	5,193	5,320
Región Autónoma del Atlántico Sur						
Río Grande Bar	2,190 (12)	2,190	809 (9)	1,079[e]	1,065 (12)	1,065
Sandy Bay Sirpi	798 (11)	870[e]	870 (12)	870	1,146 (12)	1,146
Set Net	154 (6)	310[e]	406 (12)	406	226 (12)	226
Tasbapaune	1,404 (10)	1,684[e]	2,035 (12)	2,035	2,536 (12)	2,536
Subtotal	4,546	5,054[e]	4,120	4,390[e]	4,973	4,973
TOTAL	10,366	11,077[e]	7,540	9,413[e,f]	10,166	10,166

[a] Data for the RAAN provided by C. Clark and for the RAAS by the Centro de Investigaciones y Documentación de la Costa Atlántica (CIDCA).
[b] Data for the RAAN provided by the Caribbean Conservation Corporation (CCC) and for the RAAS by CIDCA.
[c] Data for the RAAN provided by the CCC and this study, and for the RAAS by CIDCA and this study.
[d] Data obtained during the present study.
[e] Data is based on the mean number of turtles landed per month for months in which data were recorded for that year.
[f] Data is based on mean turtle landings per month for the six months prior and post the period for which no data are available.
N/A = No data are available.

APPENDIX D
PEARSON CORRELATION COEFFICIENTS FOR TEN BODY MEASUREMENTS OF GREEN TURTLES, *CHELONIA MYDAS*, HARVESTED FROM THE CARIBBEAN WATERS OF NICARAGUA

		CLN	SLN	CLT	SLT	PL	VENT	TAIL	$\bar{x}$CBASE	$\bar{x}$CLEN
SLN	♀♂	0.99 (634)								
	♀♀	0.99 (276)								
	♂♂	0.99 (282)								
CLT	♀♂	0.99 (531)	0.99 (532)							
	♀♀	0.99 (245)	0.99 (245)							
	♂♂	0.99 (246)	0.99 (247)							
SLT	♀♂	0.99 (533)	0.99 (534)	0.99 (534)						
	♀♀	0.99 (245)	0.99 (245)	0.99 (247)						
	♂♂	0.99 (248)	0.99 (249)	0.99 (247)						
PL	♀♂	0.93 (602)	0.93 (605)	0.93 (531)	0.93 (531)					
	♀♀	0.94 (276)	0.95 (276)	0.94 (247)	0.94 (247)					
	♂♂	0.91 (271)	0.91 (273)	0.91 (245)	0.92 (245)					
VENT	♀♂	0.45 (582)	0.51 (583)	0.43 (517)	0.49 (517)	0.33 (586)				
	♀♀	0.81 (272)	0.83 (272)	0.81 (243)	0.83 (243)	0.72 (276)				
	♂♂	0.82 (262)	0.86 (263)	0.81 (237)	0.84 (237)	0.70 (263)				
TAIL	♀♂	0.45 (595)	0.51 (596)	0.43 (529)	0.49 (529)	0.34 (600)	0.99 (586)			
	♀♀	0.87 (274)	0.88 (274)	0.87 (245)	0.88 (245)	0.78 (278)	0.97 (275)			
	♂♂	0.83 (272)	0.86 (273)	0.82 (246)	0.85 (246)	0.71 (274)	0.99 (264)			
$\bar{x}$CBASE	♀♂	0.43 (554)	0.49 (555)	0.42 (516)	0.48 (516)	0.34 (559)	0.94 (542)	0.95 (556)		
	♀♀	0.82 (253)	0.82 (253)	0.83 (234)	0.83 (234)	0.78 (257)	0.72 (253)	0.76 (255)		
	♂♂	0.78 (261)	0.81 (262)	0.78 (246)	0.81 (246)	0.69 (262)	0.92 (252)	0.93 (263)		
$\bar{x}$CLEN	♀♂	0.29 (456)	0.34 (547)	0.27 (508)	0.33 (508)	0.20 (551)	0.92 (534)	0.92 (548)	0.93 (553)	
	♀♀	0.48 (250)	0.50 (250)	0.51 (231)	0.52 (231)	0.46 (254)	0.54 (250)	0.52 (252)	0.67 (254)	
	♂♂	0.75 (257)	0.78 (258)	0.74 (242)	0.76 (242)	0.66 (258)	0.90 (248)	0.91 (259)	0.92 (260)	
WT	♀♂	0.95 (600)	0.95 (601)	0.95 (534)	0.95 (534)	0.92 (604)	0.40 (588)	0.40 (602)	0.38 (561)	0.23 (553)
	♀♀	0.96 (276)	0.96 (276)	0.96 (247)	0.96 (247)	0.94 (280)	0.82 (276)	0.87 (278)	0.83 (257)	0.50 (254)
	♂♂	0.94 (273)	0.95 (274)	0.94 (247)	0.94 (247)	0.89 (274)	0.84 (264)	0.84 (275)	0.79 (264)	0.75 (260)

Turtles were landed at Puerto Cabezas, Nicaragua from November 1993 to January 1995. Correlation coefficients are followed by sample sizes in parentheses.
CLN = minimum (notch-to-notch) curved carapace length, SLN = minimum (notch-to-notch) straight carapace length, CLT = maximum (tip-to-tip) curved carapace length, SLT = maximum (tip-to-tip) straight carapace length, PL = plastron length, VENT = plastron to vent length, TAIL = tail length, $\bar{x}$CBASE = mean basal area of anterior claw/turtle, $\bar{x}$CLEN = mean anterior claw length/turtle, and WT = body mass.
For all correlations $P < 0.0001$. See text for a detailed description of measurements.

APPENDIX E
REGRESSION ANALYSIS OF CURVED CARAPACE LENGTH (CLN) AGAINST NINE OTHER BODY MEASUREMENTS OF HARVESTED GREEN TURTLES, *CHELONIA MYDAS*, BY SEX FROM THE CARIBBEAN WATERS OF NICARAGUA

Test for significant difference of regression by sex							Simple linear regression equation						
X	Y	F Interc. Slope	DF	n ♀ ♂	P= Interc. Slope		a	b	n	r^2	F	DF	P <
							For pooled sexes						
CLN	SLT	0.95 2.79	3,489	245 248	0.33 0.096		1.31	0.95	529	0.98	27494	1,527	0.0001
CLN	log PL	0.06 0.01	3,543	276 271	0.81 0.91		3.38	0.0097	598	0.86	3726	1,596	0.0001
CLN	log WT	0.07 0.16	3,545	276 273	0.79 0.69		1.30	0.034	596	0.92	6859	1,594	0.0001
							For separate sexes						
CLN	CLT	10.98 11.23	3,487	245 246	0.0010 0.0009	♀	0.27	1.02	245	0.99	30031	1,243	0.0001
						♂	3.07	0.99	246	0.99	21066	1,244	0.0001
CLN	SLN	8.38 14.28	3,554	276 282	0.0039 0.0002	♀	-0.48	0.95	276	0.99	31783	1,274	0.0001
						♂	-2.78	0.98	282	0.99	21376	1,280	0.0001
CLN	log VENT	83.71 150.60	3,530	272 262	0.0001 0.0001	♀	0.53	0.023	272	0.66	536	1,270	0.0001
						♂	-1.21	0.048	262	0.68	624	1,260	0.0001
CLN	log TAIL	91.28 166.64	3,542	274 272	0.0001 0.0001	♀	0.90	0.023	274	0.75	840	1,272	0.0001
						♂	-0.76	0.047	272	0.69	656	1,270	0.0001
CLN	log $\bar{x}$CBASE	71.65 127.75	3,510	253 261	0.0001 0.0001	♀	-0.44	0.027	253	0.66	501	1,251	0.0001
						♂	-2.48	0.056	261	0.62	533	1,259	0.0001
CLN	log $\bar{x}$CLEN	75.31 135.79	3,503	250 257	0.0001 0.0001	♀	-1.23	0.014	250	0.23	76	1,248	0.0001
						♂	-3.53	0.048	257	0.56	391	1,255	0.0001

Table adapted from Limpus et al. 1994a. Turtles were landed at Puerto Cabezas, Nicaragua from November 1993 to January 1995. Male and female measurements were pooled when there was no significant difference in the slope and intercept between the sexes.
CLN = minimum (notch-to-notch) curved carapace length, SLN = minimum (notch-to-notch) straight carapace length, CLT = maximum (tip-to-tip) curved carapace length, SLT = maximum (tip-to-tip) straight carapace length, PL = plastron length, VENT = plastron to vent length, TAIL = tail length, $\bar{x}$CBASE = mean basal area of anterior claw/turtle, $\bar{x}$CLEN = mean anterior claw length/turtle, and WT = body mass.

APPENDIX F
MINIMUM NUMBER OF HAWKSBILL, *ERETMOCHELYS IMBRICATA*; LOGGERHEAD, *CARETTA CARETTA*; AND LEATHERBACK, *DERMOCHELYS CORIACEA*, TURTLES REPORTED CAPTURED AND/OR HARVESTED IN THE CARIBBEAN WATERS OF NICARAGUA FROM 1991 TO 1996

Región Autónoma del Atlántico Norte	*Eretmochelys imbricata*						*Caretta caretta*				*Dermochelys coriacea*		
	RECORDED						REC	EST TOT	RECORDED		RECORDED		
	1991[a]	1992[a]	1993[a]	1994	1995	1996	1994		1995	1996	1994	1995	1996
Awastara	N/A	N/A	4	25	14	6	33 (4)	99[b]	92	327	0	0	0
Dakra	N/A	N/A	N/A	14	1	11	8 (4)	24[b]	7	34	0	3	0
Sandy Bay	N/A	N/A	N/A	22	9	12	15 (4)	45[b]	27	21	0	0	0
Other[c]	N/A	N/A	2	13	3	0	1	1	6	3	0	0	0
Subtotal	N/A	N/A	6	74	27	29	57	169[b]	132	385	0	3	0
Región Autónoma del Atlántico Sur													
Río Grande Bar	N/A	1	N/A	2	3	6	3	3	7	8	0	0	0
Sandy Bay Sirpi	3	1	13	5	10	5	N/A	N/A	11	18	0	0	0
Set Net	N/A	N/A	N/A	2	12	4	1	1	10	9	0	0	0
Tasbapaune	N/A	N/A	N/A	3	57	9	0	0	9	63	0	1	0
Subtotal	3	2	13	12	82	24	4	4	37	98	0	1	0
TOTAL	3	2	19	86	109	53	61	173[b]	169	483	0	4	0

Numbers in parentheses are months.
REC = Recorded, EST TOT = Estimated Total, N/A = No data are available.
[a] Data for 1991 - 1993 were provided by the Centro de Investigaciones y Documentación de la Costa Atlántica.
[b] Data is based on the mean number of turtles landed per month for months in which data were recorded for that year.
[c] Includes the communities of Krukira and Pahra and mechanized fishing boats.

APPENDIX G
SUMMARY STATISTICS OF BODY SIZE PARAMETERS FOR HARVESTED HAWKSBILL, *ERETMOCHELYS IMBRICATA*, TURTLES

Body Measurements (cm)	Female			Male		Combined		
	Mean (SD)	Range	n	Mean (SD)	n	Mean (SD)	Range	n
CLN	77.8 (7.4)	67.0-85.6	5	73.7	1	77.2 (6.8)	67.0-85.6	6
SLN	73.8 (5.1)	68.2-78.2	3			73.8 (5.1)	68.2-78.2	3
CLT	84.9 (5.4)	77.6-90.5	4			84.9 (5.4)	77.6-90.5	4
SLT	78.4 (5.6)	72.5-83.7	3			78.4 (5.6)	72.5-83.7	3
PL	59.1 (4.3)	54.1-63.5	5	53.4	1	58.2 (4.5)	53.4-63.5	6
VENT	12.5 (1.8)	11.5-15.7	5	14.6	1	12.9 (1.8)	11.5-15.7	6
TAIL	17.3 (2.2)	15.2-20.9	5	18.6	1	17.5 (2.0)	15.2-20.9	6
$\bar{x}$CBASE	6.0 (0.5)	5.6-6.4	2			6.0 (0.5)	5.6-6.4	2
$\bar{x}$CLEN	1.4 (0.2)	1.3-1.6	2			1.4 (0.2)	1.3-1.6	2
WT (kg)	53.3 (13.0)	45.4-72.6	4			53.3 (13.0)	45.4-72.6	4

CLN = minimum (notch-to-notch) curved carapace length, SLN = minimum (notch-to-notch) straight carapace length, CLT = maximum (tip-to-tip) curved carapace length, SLT = maximum (tip-to-tip) straight carapace length, PL = plastron length, VENT = plastron to vent length, TAIL = tail length, $\bar{x}$CBASE = mean basal area of anterior claw/turtle, $\bar{x}$CLEN = mean anterior claw length/turtle, and WT = body mass. See text for a detailed description of measurements.

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BIOGRAPHICAL SKETCH

Cynthia Jean Lagueux was born in Minneapolis, Minnesota, on 20 July 1954. At an early age she was exposed to many cultures, customs, and languages. During her pre-school years she and her family lived in Burma (now Myanmar) for three and one-half years. During this period they had the opportunity to travel around the world twice visiting many countries. In 1972, Ms Lagueux graduated from Edison high school in Minneapolis. In 1978, she attended the University of Minnesota and completed her Bachelor of Science degree in Wildlife Management. After graduating, she worked for several years with the Minnesota Hennepin County Park Reserve District in the management and restoration of native plant and animal species.

For four years, from 1981 to 1985, she was a Peace Corps volunteer in Honduras, Central America. She worked with the Honduran Department of Recursos Naturales Renovables on a variety of wildlife and human related issues, e.g., sport hunting of migratory doves, conservation of psittacines, introduction and implementation of an environmental education program at the primary school level, and the conservation of marine turtles. Since this initial exposure to sea turtles in 1981, Ms Lagueux has worked with four of the seven sea turtle species in four countries.

Ms Lagueux's interest in human use of natural resources and their conservation prompted her to pursue a masters degree at the University of Florida. Her research was

on the use of marine turtle eggs by humans, "Olive ridley (Lepidochelys olivacea) nesting in the Gulf of Fonseca and the commercialization of its eggs in Honduras." In 1989, she graduated with a Masters of Arts from the Center for Latin American Studies in the Program in Studies for Tropical Conservation at the University of Florida. Immediately after graduating she began working towards her doctoral degree.

In 1992, as a consultant to the Miskito Cays Protected Area, she began conducting research on human use and conservation of sea turtles in Nicaragua. She returned to Nicaragua in October 1993 and conducted research on the marine turtle fishery for her doctoral dissertation. After completing her Ph.D., Ms Lagueux will continue her research on human use and conservation of marine turtles on the Caribbean coast of Nicaragua as a Research Fellow with the Wildlife Conservation Society. Along with several colleagues she hopes to establish a long-term, community-based approach to the management of the Miskitu Indian marine turtle fishery.

I certify that I have read this study and that in my opinion it conforms to acceptable standards of scholarly presentation and is fully adequate, in scope and quality, as a dissertation for the degree of Doctor of Philosophy.

Kent H. Redford, Chair
Associate Professor of Wildlife Ecology and Conservation

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Louis J. Guillette, Jr.
Professor of Zoology

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Richard E. Bodmer
Assistant Professor of Wildlife Ecology and Conservation

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George W. Tanner
Professor of Wildlife Ecology and Conservation

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Lyn C. Branch
Associate Professor of Wildlife Ecology and Conservation

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Assistant Professor of Zoology

This dissertation was submitted to the Graduate Faculty of the College of Agriculture and to the Graduate School and was accepted as partial fulfillment of the requirements for the degree of Doctor of Philosophy.

May 1998

Dean, College of Agriculture

Dean, Graduate School